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Evaluation of Uncertainty in Fraser Sockeye (*Oncorhynchus nerka*) Wild Salmon Policy Status using Abundance and Trends in Abundance Metrics

Évaluation de l'incertitude de la situation de la Politique concernant le saumon sauvage à l'égard du saumon du Fraser (*Oncorhynchus nerka*) à l'aide de l'abondance et des tendances dans les paramètres d'abondance

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This series documents the scientific basis for the evaluation of aquatic resources and ecosystems in Canada. As such, it addresses the issues of the day in the time frames required and the documents it contains are not intended as definitive statements on the subjects addressed but rather as progress reports on ongoing investigations.

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ABSTRACT

The Department of Fisheries and Oceans (DFO) Wild Salmon Policy (WSP) goal is "to restore and maintain healthy salmon populations and their habitats for the benefit and enjoyment of the people of Canada in perpetuity" (Fisheries and Oceans Canada 2005). In order to achieve this goal, the WSP outlines a number of strategies, including '*Strategy 1: standardized monitoring of wild salmon statuses'*, which is the subject of this paper. In the current paper, Fraser Sockeye (*Oncorhynchus nerka*) conservation units (CUs) from 'WSP *Action Step 1.1: the identification of conservation units'* are used to update '*Action Step 1.2: the development of criteria to assess CUs and identify benchmarks to represent biological statuses'*, and to address '*Action Step 1.3: CU status assessment'*, for the 22 current CUs and two *de novo* 'CUs'. Using a previously developed toolkit for CU status assessment (Holt et al. 2009; Holt 2009), *abundance* benchmarks (unique to each CU) were estimated for each CU with stock-recruitment data, and *trends in abundance* upper and lower benchmarks (common across all CUs) were modified to apply to Fraser Sockeye. These benchmarks were used to delineate the three WSP biological status zones (Red, Amber, and Green). *Abundance* benchmarks were estimated across a range of stock-recruitment models, including the standard Ricker model that assumes constant productivity and other Ricker model forms that assume time varying productivity. Consideration of time varying productivity in the estimation of *abundance* benchmarks was important since most Fraser Sockeye CUs have exhibited systematic declines in productivity over recent decades (Grant et al. 2011) and extirpation risk can increase when a CUs productivity is linearly decreasing or low (Holt 2009; Holt and Bradford 2011). *Abundance* benchmarks were also estimated across a range of probability levels to reflect uncertainty in the estimation process. Estimates of a CU's spawner abundances at maximum juvenile production (S_{max}) were also updated and used as carrying capacity priors in Ricker models, where available and appropriate. In the evaluation of status using the *abundance* metric, both the geometric and arithmetic means of the recent CU abundance were compared against benchmarks. Since multiple metrics (one *abundance* and three *trends in abundance* metrics, depending on the CU) and uncertainty in *abundance* benchmarks are presented in the current paper, statuses for a single CU can comprise all three WSP status zones. Status integration will be explored in future processes and publications.

RÉSUMÉ

L'objectif de la Politique concernant le saumon sauvage (PSS) du ministère des Pêches et des Océans (MPO) est « de rétablir et de maintenir en bon état de santé les populations de saumon et leur habitat, pour le bénéfice et le plaisir perpétuels des citoyens du Canada » (Pêches et Océans Canada, 2005). Pour atteindre cet objectif, la PSS donne un aperçu de diverses stratégies, dont la « Stratégie 1 Surveillance normalisée de l'état du saumon sauvage », faisant l'objet du présent document. Dans le présent document, les unités de conservation (UC) du saumon du Fraser (*Oncorhynchus nerka*) de l'« étape 1.1 de la PSS : Repérage des unités de conservation » sont utilisées pour mettre à jour l'« étape 1.2 : Élaboration des critères d'accès aux UC et des indices de référence pour représenter l'état biologique », et pour aborder l'« étape 1.3 : Contrôle et évaluation de l'état des UC », pour les 22 UC actuelles et les deux nouvelles « UC ». À l'aide d'une trousse d'outils déjà élaborée pour l'évaluation de la situation de l'UC (Holt *et coll.* 2009; Holt 2009), les indices de référence pour l'*abondance* (unique à chaque UC) ont été estimés pour chaque UC en utilisant les données de stock-recrutement. Les indices de références supérieurs et inférieurs des tendances en matière d'abondance (communs dans l'ensemble des UC) ont été modifiés afin qu'ils s'appliquent au saumon du Fraser. Ces indices de références ont été utilisés pour délimiter les trois zones de situation biologique de la PSS (rouge, ambre et vert). Les indices de référence sur l'abondance ont été estimés dans une gamme de modèles de stock-recrutement, dont le modèle standard de Ricker qui suppose une productivité stable et les autres formes du modèle de Ricker qui supposent une productivité temporisée. La prise en compte de la productivité temporisée dans l'estimation des indices de référence de l'abondance était importante étant donné que la plupart des UC du saumon du Fraser ont affiché une diminution générale de la productivité au cours des dernières décennies (Grant *et coll.* 2011) et le risque de disparition d'un endroit donné peut augmenter lorsque la productivité d'une UC diminue de façon linéaire ou est faible (Holt 2009; Holt et Bradford 2011). Des indices de référence de l'abondance ont également été estimés dans une gamme de niveaux de probabilité afin de tenir compte de l'incertitude dans le processus d'estimation. Les estimations des abondances de reproducteurs d'une UC à la production juvénile maximum (S_{max}) ont également été mises à jour et utilisées comme capacité limite antérieure dans les modèles de Ricker, lorsqu'elles sont offertes et que cela convient. Lors de l'évaluation de la situation se servant du paramètre de l'abondance, les moyennes géométrique et arithmétique de l'abondance récente de l'UC ont été comparées aux indices de références. Étant donné que de nombreux paramètres (un paramètre sur l'abondance et trois paramètres sur les tendances en matière d'abondance, selon l'UC) et l'incertitude dans les indices de référence de l'abondance ont été présentés dans le présent document, les situations d'une seule UC peuvent comprendre les trois zones de la PSS. Le regroupement de la situation sera examiné dans de prochains processus et prochaines publications.

INTRODUCTION

The Department of Fisheries and Oceans (DFO) Wild Salmon Policy (WSP) goal is "to restore and maintain healthy salmon populations and their habitats for the benefit and enjoyment of the people of Canada in perpetuity" (Fisheries and Oceans Canada 2005). In order to achieve this goal, the WSP outlines a number of strategies, including '*Strategy 1: standardized monitoring of wild salmon statuses*, which is the subject of this paper. Under this strategy, this paper uses Fraser Sockeye conservation units (CUs) identified through 'WSP *Action Step 1.1: the identification of conservation units (CUs)'* to update '*Action Step 1.2: the development of criteria to assess CUs and identify benchmarks to represent biological statuses*, and to address '*Action Step 1.3: CU status assessment*', for the 22 current CUs and two *de novo* 'CUs' (see Appendix 6 for the original request for Science Advice). Since several metrics are used to assess status for each CU, and uncertainties in *abundance* benchmarks are considered in the current assessment, the resulting statuses for each CU can include a combination of the three WSP biological status zones (Red, Amber and Green)(Figure 1). Integration of divergent statuses for a CU into a final single status will be addressed in subsequent processes and publications.

Methodology for the identification of CUs and a consequent list of draft CUs for salmon stocks in the Pacific Region (WSP Action Step 1.1) was presented in Holtby and Ciruna (2007). The CU list is not static, but is subject to change as new analyses and data become available. The current paper presents the Fraser Sockeye CU list last revised on August 16, 2011 (Table 1) (B. Holtby, DFO, pers. comm.). This list includes 22 current CUs and two *de novo* (considered new to due hatchery transplantation) 'CUs'. Two of the 22 current CUs, Chilko-S and Chilko-ES, cannot be assessed independently since escapement data for these CUs are aggregated. In addition to the current and *de novo* CUs, there are six CUs that require further research to confirm that they are CUs, based on Holtby and Ciruna (2007) methodology, or to determine whether or not they have been extirpated. There are also eight extirpated (locally extinct) CUs and six CUs that have been removed from previous versions of the CU list due to recent additional considerations (Holtby, DFO, pers. comm.). The corresponding stock names for all of these CUs, used commonly in fisheries management processes, are presented in Table 2.

Coincidental to the identification of CUs, methodology for the assessment of Pacific salmon CU status (WSP Action Step 1.2) was presented in two recent papers (Holt 2009; Holt *et al.* 2009). Classes of indicators recommended for the assessment of status include *abundance*, *trends in abundance*, *fishing mortality*, and *distribution* (Figure 2). Within each class of indicator, more than one metric can be used to assess status (Figure 2). For each metric, lower and upper benchmarks delineate, respectively, the Red to Amber and the Amber to Green WSP biological status zones (Figures 1 & 2). To meet the definition specified in the WSP, the lower benchmark is set at a level that ensures there is a substantial buffer between the benchmark and the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) classification of 'endangered'. Conservation units in the Green zone are considered at low risk for extinction, and could sustain on average, maximum annual catches given existing environmental conditions. Although changes in status are intended to inform management decision making, on their own they are not prescriptive.

Since a relatively complete time series of escapement and recruitment data exists for a large number of Fraser Sockeye CUs, the classes of indicators explored in this assessment of status include *abundance* and *trends in abundance* (WSP Action Step 1.2). The *fishing mortality* class of indicator differs from the remaining three (Figure 2), as it reflects a threat to the CU rather than an intrinsic property of the CU, and is typically used only when abundance data are not available (Holt et al. 2009). For Fraser Sockeye, the *fishing mortality* class of indicator is not used to assess CU status, since abundance data are available. Further, a recent DFO workshop concluded that further research on *fishing mortality* benchmarks and their usefulness in status evaluation is required prior to their use. The *distribution* class of indicator is also not assessed in the current paper because escapement enumeration methods generally do not provide the flexibility to assess distributional changes through time. Also, additional work is required to determine how distribution affects a CU's extirpation risk and status. Therefore, only the *abundance* and *trends in abundance* metrics are considered for Fraser Sockeye CU status assessments.

Table 1. The 22 current CUs, two de novo (new) 'CUs', six CUs that require further research to validate, eight extirpated CUs (two of which are being studied for recovery potential), and five CUs that were removed from previous versions of the CU list.

Current	*De Novo* (New)	Validation Required	Extirpated	Removed (no longer CUs)
1 Anderson-Seton-ES	1 North Barriere-ES[2]	1 Cariboo-S (extirpated?)	1 Adams-ES	1 Boundary Bay
2 Bowron-ES	2 Seton-L	2 Francois (First Run)-ES (extirpated?)	2 Alouette-ES	2 Carpenter Lake
3 Chilko-ES[1]		3 Francois (Second Run)-ES (extirpated?)	3 Coquitlam-ES	3 Fraser Canyon
4 Chilko-S[1]		4 Indian/Kruger-ES (extirpated?)	4 Fraser-ES	4 Hayward Lake
5 Chilliwack-ES		5 Middle Fraser (River-Type)	5 Kawkawa-L	5 Thompson (River-Type)
6 Cultus-L		6 Upper Fraser (River-Type)	6 Momich-ES	6 Stuart-Early Stuart
7 Francois-Fraser-S			7 North Barriere-ES[2]	
8 Harrison (D/S)-L			8 Seton-S	
9 Harrison (U/S)-L				
10 Harrison River (River-Type)				
11 Kamloops-ES[2]				
12 Lillooet-Harrison-L				
13 Nadina-Francois-ES (new-mixed CU)				
14 Nahatlach-ES				
15 Pitt-ES				
16 Quesnel-S				
17 Shuswap-ES				
18 Shuswap Complex-L				
19 Takla-Trembleur-EStu				
20 Takla-Trembleur-Stuart-S				
21 Taseko-ES				
22 Widgeon (River-Type)				

1. **Chilko-ES and Chilko-S** are aggregated for CU assessment purposes; these data sets cannot be disaggregated.
2. **Kamloops-ES** does not include extirpated populations upstream of previous dam on Barriere River which are now part of the North Barriere-ES (extirpated) CU; hatchery transplants in the North Barriere system, after dam removal, has produced new growing populations, now identified as the North Barriere-ES (*De Novo*) CU.

Holt et al. (2009) and Holt (2009) have developed a framework for estimating benchmarks for *abundance* indicators (unique to each CU) and identified benchmarks for *trends in abundance* indicators (common across all CUs). For *abundance* lower and upper benchmarks, Holt (2009) recommended using, respectively, S_{gen} (the spawner abundance that would result in recovery to maximum sustained yield (S_{MSY}) in one generation) and 80% S_{MSY}. Simulation modelling results indicated that, compared to other benchmarks, using S_{gen} (estimated using a Ricker stock-recruitment model cast in a Bayesian framework) as a lower benchmark was associated with a relatively low probability (<25%) of extirpation over 100 years for populations under equilibrium abundances greater than 15,000 spawners. This S_{gen} lower benchmark also had a relatively high probability (>75%) of recovery to the spawning abundance necessary to provide maximum sustained yield (S_{MSY}) in three generations (Holt 2009; Holt & Bradford 2011). The S_{gen} lower benchmark was also more robust to uncertainties in productivity compared to other benchmarks evaluated (Holt 2009).

Table 2. The 22 current CUs and two de novo (new) 'CUs' and their corresponding stock name (commonly used in fisheries management processes).

CU Name	Corresponding Stock Name
Current	
1 Anderson-Seton-ES	Gates
2 Bowron-ES	Bowron
3 Chilko-ES[1]	Chilko
4 Chilko-S[1]	Chilko
5 Chilliwack-ES	Miscellaneous Early Summers
6 Cultus-L	Cultus
7 Francois-Fraser-S	Stellako
8 Harrison (D/S)-L	Miscellaneous Lates
9 Harrison (U/S)-L	Weaver
10 Harrison River (River-Type)	Harrison
11 Kamloops-ES	Raft and miscellaneous Early Summers
12 Lillooet-Harrison-L	Birkenhead
13 Nadina-Francois-ES (new-mixed CU)	Nadina
14 Nahatlach-ES	Miscellaneous Early Summers
15 Pitt-ES	Pitt
16 Quesnel-S	Quesnel
17 Shuswap-ES	Scotch and Seymour and miscellaneous Early Summers
18 Shuswap Complex-L	Late Shuswap
19 Takla-Trembleur-EStu	Early Stuart
20 Takla-Trembleur-Stuart-S	Late Stuart
21 Taseko-ES	Miscellaneous Early Summer
22 Widgeon (River-Type)	Miscellaneous Lates
***De Novo* (New)**	
1 North Barriere-ES	Fennell and miscellaneous Early Summers
2 Seton-L	Portage

1. **Chilko-ES and Chilko-S** are aggregated for CU assessment purposes; these data sets cannot be disaggregated.

Holt et al. (2009) recommended using the Ricker stock-recruitment model using a Bayesian approach to estimate lower (S_{gen}) and upper (80% S_{msy}) *abundance* benchmarks. Simulation modelling has shown that Ricker stock-recruitment model benchmarks are relatively robust to extirpation risk and recovery (Holt 2009), and do not vary considerably when the "true" underlying stock-recruitment relationship follows a Larkin stock-recruitment model with highly cyclic population dynamics (Holt and Bradford 2011). Therefore, only Ricker model forms were used to estimate CU benchmarks for *abundance* metrics in the current paper. In addition to the standard Ricker stock-recruitment model (using the full stock-recruitment time series), which assumes constant productivity, Ricker model forms that assume time varying productivity were also used to estimate *abundance* benchmarks. This was an important consideration for Fraser Sockeye CUs, as most of these CUs have exhibited persistent declines in productivity in recent decades (Grant et al. 2011; Appendix 3, Figures 1 c & d). Estimates of the spawner abundances at maximum juvenile production (S_{max}) for CUs were also updated in the current paper, and were used as Bayesian priors for carrying capacity in the Ricker models, where available and appropriate.

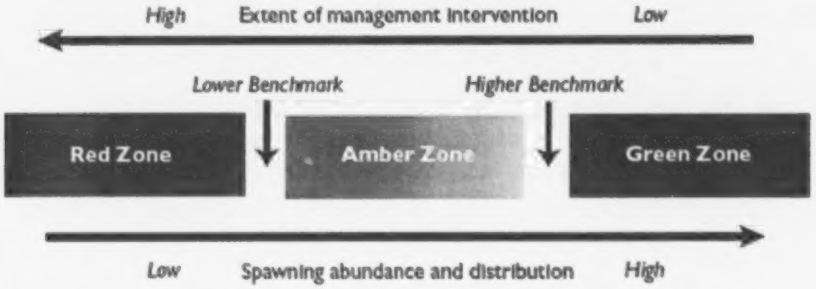
High
Extent of management intervention
Low
Lower Benchmark
Higher Benchmark
Red Zone
Amber Zone
Green Zone
Low
Spawning abundance and distribution
High

Figure 1. Wild Salmon Policy status zones (Red, Amber, and Green) delineated by lower and upper benchmarks. Increasing spawner abundance is inversely related to the extent of management intervention. Reprinted from Fisheries and Oceans Canada (2005).

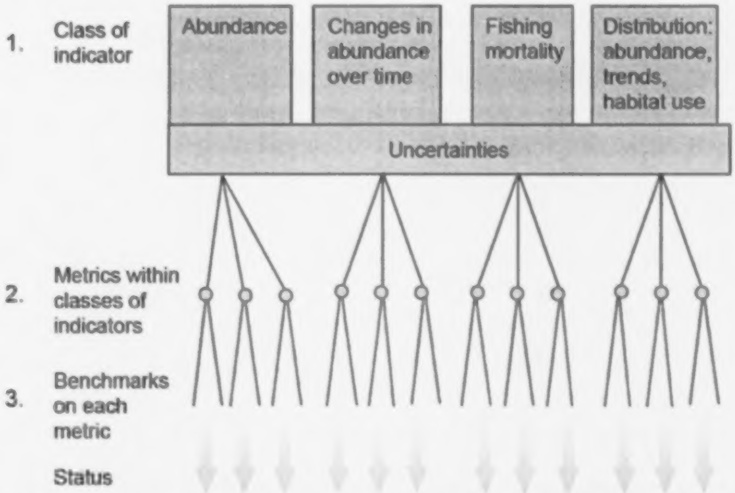
1. Class of indicator
Abundance
Changes in abundance over time
Fishing mortality
Distribution: abundance, trends, habitat use
Uncertainties
2. Metrics within classes of indicators
3. Benchmarks on each metric
Status

Figure 2. Hierarchy for the assessment of biological status of WSP CUs, including 1) four classes of indicators, 2) quantifiable metrics within each indicator class, and 3) benchmarks on each metric. Reprinted from Holt et al. (2009).

Trends in abundance metrics and associated benchmarks (common across all CUs) were presented in Holt et al. (2009). Results from a recent study (Porszt 2009) and updated analyses in the current paper, compared the relative performance of various *trends in abundance* metrics in correctly identifying 'true' status. These results, in addition to the Holt et al. (2009) *trends in abundance* metric recommendations, were used to select *three trends in abundance* metrics for the assessment of Fraser Sockeye CU status: the ratio of the current abundance to the long-term average (long-term trends in abundance), the linear change in abundance over the past three generations (recent trends in abundance), and the probability that this recent change in abundance is below the lower benchmark for this metric (recent trends in abundance). Benchmarks for each of these metrics (common across all CUs) were modified from Holt et al. (2009) and are described in the methods section of the current paper.

For each CU, statuses were evaluated across the range of metrics (one *abundance* and three *trends in abundance*) considered. Further, abundance statuses were evaluated across the range of estimated *abundance* benchmarks (across different model forms and probability levels), which compared two different calculations of the average recent abundance (geometric versus arithmetic mean) against paired sets of upper and lower benchmarks. This evaluation of uncertainty in *abundance* benchmarks and the use of multiple metrics to evaluate status in the current assessment resulted in a suite of statuses for each Fraser Sockeye CU that could comprise all three WSP status zones. These results provide the foundation required for developing a final single status for each Fraser Sockeye CU, which will be addressed in subsequent processes and publications.

The objectives of the current paper are to present the following information:

A) background on Fraser Sockeye life-history, population trends and structure, and threats;

B) an updated list of Fraser Sockeye CUs, including 22 current CUs and two *de novo* 'CUs', six CUs that require further research to validate, eight extirpated CUs (two of which are currently being studied for recovery potential), and six CUs that have been removed from the previous CU list (WSP Strategy 1, Action Step 1.1);

C) for all 24 assessable CUs (22 current and two *de novo* CUs), escapement sites, history, escapement time series, productivity, abundance and trends in abundance are reported; background is provided for all remaining CUs (six validation required, eight extirpated, and six removed from the current CU list);

D) updated estimates of spawner abundances at maximum juvenile production (S_{max}) organized by juvenile rearing lake; these data are used as Ricker model Bayesian prior values of carrying capacity (Ricker '*b*' parameter) used to estimate *abundance* benchmarks for some CUs (where available and appropriate);

E) evaluation of uncertainty in *abundance* benchmarks for each CU with stock-recruitment data and modified Holt et al. (2009) *trends in abundance* benchmarks (WSP Strategy 1, Action Step 1.2);

F) for each CU, evaluation of status for each metric, including a range of *abundance* benchmarks that reflect both structural and stochastic uncertainty (addresses WSP Strategy 1, Action Step 1.3).

FRASER SOCKEYE BACKGROUND

SPECIES CLASSIFICATION AND DESCRIPTION

Sockeye salmon are one of the seven species of Pacific salmon. Sockeye salmon develop secondary sexual characteristics as they return to the spawning grounds, similar to other Pacific Salmon. Adult Sockeye spawning characteristics include bright red body coloration, olive green heads and tails, and an elongated snout. Spawning Sockeye are sexually dimorphic; males are distinguished from females by a fleshy back hump located between their head and dorsal fin, and a curved upper jaw with protruding canine-like teeth. The juvenile smolt stage is characterized by oval parr marks of irregular heights that largely occur above the lateral line (Pollard et al. 1997). In their ocean phase, Sockeye are silver-blue in coloration, have no spots on their back or tail, are slim and tubular, and can range in weight from 2.2 to 3.1 kg (maximum: 6.3 kg). More detailed descriptions of Fraser Sockeye are available (Foerster 1968; Hart 1973; Burgner 1991).

FRASER WATERSHED

The Fraser River supports the largest abundance of Sockeye salmon in the world for a single river (Northcote and Larkin 1989), due to its length (1,600 km), watershed size (223,000 km^2), and lake nursery area (2,500 km^2) (Figure 3). Over fifty percent of all salmon production in British Columbia (over sixty-five percent for Sockeye) occurs in the Fraser watershed. From its headwaters in the Rocky Mountains, the Fraser River follows the Rocky Mountain Trench to the Interior Plateau. It continues south to the Coast Mountains and drains from a broad floodplain into the Strait of Georgia. The Lower Fraser watershed and the Upper Fraser watershed are divided by the narrow Hells Gate canyon (Figure 3). Within the Fraser watershed there are hundreds of tributaries, steams, marshes, bogs, swamps, sloughs, and lakes. As a result of this large system, Fraser Sockeye spawning migration can range from tens to thousands of kilometres (Figure 3).

FRASER SOCKEYE LIFE HISTORY

Overview

The dependence of Sockeye salmon on specific lakes for juvenile habitat has resulted in a greater variety of life history patterns, relative to other species of Pacific salmon. Two key life-history types of Sockeye salmon include anadromous Sockeye (*O. nerka* that have both freshwater and marine phases) and kokanee (*O. nerka* that spend their entire life-cycle in freshwater). These two forms of *O. nerka* have diverged genetically (Taylor *et al.* 1996; Taylor *et al.* 1997; Foote *et al.* 1999; Craig and Foote 2001) and ecologically (Foote *et al.* 1999; Wood *et al.* 1999). These two Sockeye forms likely do not interbreed due to differences in spawning times and anadromous female Sockeye mate selection, which favours the larger anadromous males over the smaller non-anadromous males. The current paper focuses on the anadromous form of Sockeye Salmon, that spawn (and subsequently die) as adults in freshwater, incubate as eggs in gravel in the freshwater environment, and either migrate to the ocean shortly after gravel emergence as fry, or migrate to the ocean as smolts after rearing in freshwater lakes for one to three years. Anadromous Sockeye spend an additional one to three years rearing in the ocean as juveniles before they return to spawn.

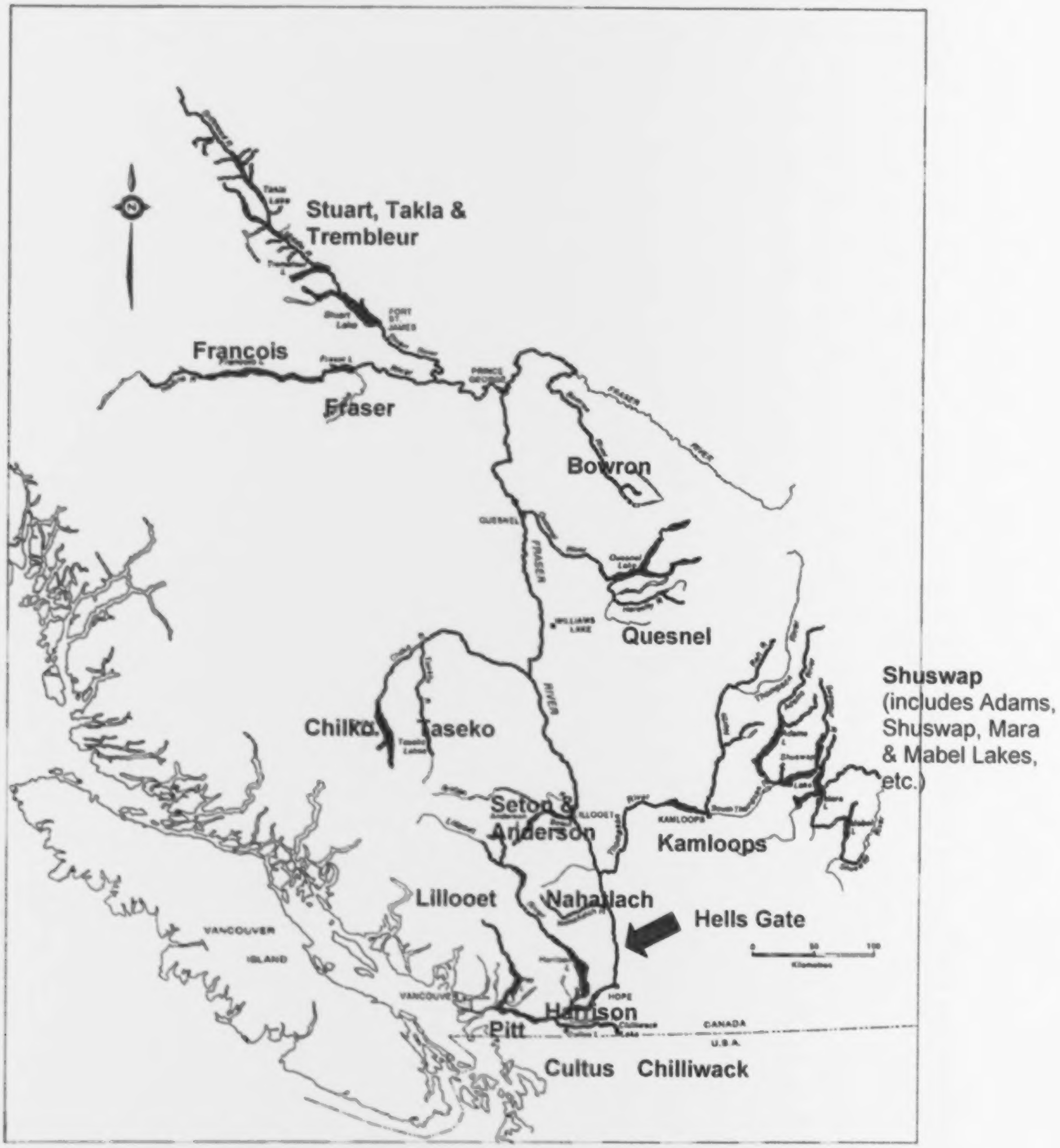
Stuart, Takla &
Trembleur
Francois
Fraser
FORT ST. JAMES
PRINCE GEORGE
FRASER RIVER
Bowron
QUESNEL
FRASER
Quesnel
WILLIAMS LAKE
RIVER
Shuswap
(includes Adams,
Shuswap, Mara
& Mabel Lakes,
etc.)
Chilko
Taseko
Seton &
Anderson
LILLOOET
KAMLOOPS
Kamloops
Lillooet
Nahatlach
Hells Gate
Kilometres
VANCOUVER
ISLAND
VANCOUVER
HOPE
Harrison
Pitt
CANADA
U.S.A.
Cultus
Chilliwack

Figure 3. Sockeye salmon freshwater distribution in the Fraser River watershed with key CU lakes identified (blue text). Heavy black indicates locations of Sockeye spawning. Hells Gate is indicated on map (red arrow and text), as this location presents a challenge to adult Sockeye upstream migration during high flows in the Spring/Early Summer periods.

Age-Structure

Depending on the population, adult Fraser Sockeye salmon can range in age from three to six years, spending their first one to three winters in freshwater and their last one to three winters in the marine environment. Most Fraser Sockeye, however, return to spawn as four-year-old fish (age 4_2 based on the Gilbert Rich ageing convention) (~80% of the total adult recruitment age composition), after spending two winters in the freshwater followed by two winters in the marine environment. A smaller proportion (~20% of the total adult age composition) of Sockeye spends one extra winter in the marine environment (age-5: 5_2). Fraser Sockeye are also comprised of a small component of three-year-old fish (typically called jacks/precocious males or jills/precocious females, although jills are far less common) that return to spawn after only one year in the ocean (age 3_2). One exception to this age composition occurs in Pitt River Sockeye (Pitt-ES CU), which predominantly return as five-year-old fish (~65% 5_2 out of the total sum of 4_2 & 5_2's). For all CUs, there can be a very small proportion (1.6% out of the total recruitment) of fish that spend three winters in freshwater and varying lengths of time in the marine environment (ages: 4_3,5_3,6_3). In recent years (1980 to present), maturation appears to have delayed, as returns are comprised of increasing proportions of four year olds relative to three year olds and five year olds relative to four year olds (Holt and Peterman 2004; Grant et al. 2010). Overall, however, four-year-olds continue to dominate recruitment for most stocks.

A major life-history variant occurs in the Harrison River (Harrison River (River-Type) CU) and Widgeon Slough (Widgeon (River-Type) CU). Sockeye in these CUs are comprised of age-3 (3_1) and age-4 (4_1) fish that do not rear in freshwater lakes as juveniles. For the Harrison River (River-Type) CU, the proportion of recruits that return as three or four year olds is highly variable, with higher percentages of age-4 fish (~65%) returning during odd years when pink salmon are also spawning in this system (Grant *et al.* 2010).

Adult Return Migration, Spawning, and Freshwater Residence

Fraser Sockeye return from the North Pacific to the Strait of Georgia via either the northern Johnstone Strait or the southern Juan de Fuca Strait route. The proportion travelling through Johnstone Strait varies from 2 to 80% (Groot and Cooke 1987), and is affected by El Niño events that result in higher diversion rates through Johnstone Strait due to warmer water flows from the south (Groot and Quinn 1987).

The natural homing of Sockeye to their spawning areas is precise in both timing and location, more so than in other species of Pacific Salmon (Burgner 1991). Return migration timing is related to temperature regimes in the egg incubation areas, to ensure appropriate development and emergence timing of eggs and fry (Miller and Brannon 1982). The spawning period for Fraser Sockeye can range from July to October, and adults typically cease feeding as they enter the freshwater system (Burgner 1991; COSEWIC 2003).

Adult Sockeye usually spawn in rivers, streams and along lake foreshores. Typical of the genus *Oncorhynchus*, eggs are deposited in nests constructed by the female, fertilized by a male or an opportunistic precocious male, and then subsequently covered with gravel by the female. Nests are dug in gravel that ranges in size from coarse sand to large angular rubble and boulders. Water depth ranges from 0.1 meters in small streams to over 30 m in lakes; water temperature ranges from 2 to 8° C. Eggs incubate in the gravel through the winter, with incubation duration and the timing of fry emergence from the gravel (mid-April to mid-May) mediated by ambient temperatures (Burgner 1991). Following fry emergence, the progeny of river spawners migrate in schools to the lake, where they move along the shoreline in shallow water before progressively moving offshore (Morton and Williams 1990). In Cultus Lake, the progeny of shore spawners immediately migrate into deep water (Brannon 1972). Juveniles rear in the lake for one to two winters after gravel emergence. In most cases, fry rear in lakes immediately downstream of their natal spawning streams, but exceptions have been documented. For example, fry from Gates Sockeye (Anderson-Seton-ES CU) initially enter Anderson Lake to rear. However, a variable and often substantial proportion of fry continue their migration through this lake to rear further downstream in Seton Lake (Geen and Andrew 1961). Similarly, an occasionally large proportion of fry from the Birkenhead River (Lillooet-Harrison-L CU) initially enter Lillooet Lake, then migrate through this lake to rear in Harrison Lake (Cave 1988). In both cases, the growth of fry in the second lake appears to be higher than in the original nursery lake (J. Hume, data on file). In addition, instead of downstream migration, some Fraser Sockeye fry migrate upstream to their rearing lake (e.g. Harrison (U/S)-L and Chilko-S).

Smolt Outmigration and Marine Residence

Following a period of either egg incubation and fry lake rearing (lake-type Sockeye) or egg incubation only (river-type Sockeye), Fraser Sockeye juveniles migrate downstream in the spring (April to June). This migration is relatively quick as the juveniles move downstream from their rearing lake (or incubation gravel), through the Fraser River and Fraser estuary, and into the Strait of Georgia (Healey 1980; Tucker *et al.* 2009; Welch *et al.* 2009). Upon entry into the Strait of Georgia, most Sockeye migrate northward through Johnstone Strait and along the continental shelf, before entering the North Pacific sometime between the fall and winter period (Tucker *et al.* 2009; Welch *et al.* 2009) (Figure 4). Stellako (Francois-Fraser-S CU) and Stuart (Takla-Trembleur-Stuart-S, Takla-Trembleur-EStu CUs) Sockeye appear to leave the continental shelf somewhat earlier (in the fall) than all other stocks (Tucker et al. 2009). After leaving the continental shelf, juvenile Fraser Sockeye salmon in the North Pacific high seas are widely distributed in the Gulf of Alaska, between 48°N and 60°N and 125°E to 170°E (Forrester 1987) (Figure 4). Harrison River (River-Type) (i.e. Harrison River) Sockeye have unique ocean migration timing and migration routes. After emergence from the spawning gravel, Harrison Sockeye rear in sloughs for a few months prior to their downstream migration, and, as a result, enter the Strait of Georgia a few months after all other Fraser Sockeye (Birtwell et al. 1987). Also, unlike all other Fraser Sockeye, Harrison Sockeye rear in the Strait of Georgia for up to six months prior to migrating through the Southern Juan de Fuca Strait to the Gulf of Alaska (Taylor *et al.* 1997; Tucker *et al.* 2009; R. Beamish, DFO, pers. comm.).

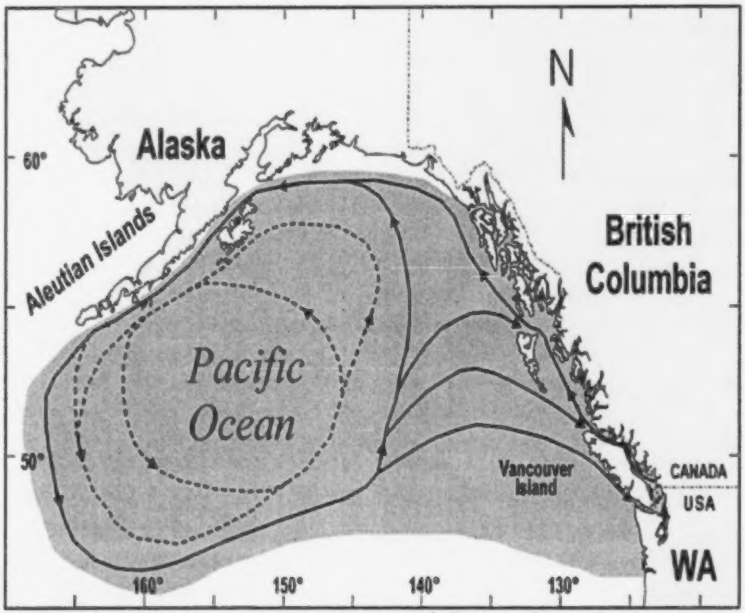

Figure 4. The open ocean migration pattern of Fraser River Sockeye salmon; grey area is overall distribution, black lines are main routes and dashed lines indicate other areas covered (modified from French et al. (1976) & Healy (2002); reprinted from Johannessen and Ross (2002)).

FRASER SOCKEYE POPULATION SIZE AND TRENDS

Cycles and Escapement, Catch and Return Trends

Fraser Sockeye predominantly return to spawn as four-year old fish, and for some of the large lakes in the Fraser River watershed, persistent four-year cycles of Sockeye abundance occur (cyclic dominance). Patterns of cyclic dominance in abundance trends varies amongst CUs and can include one very large (dominant) cycle, followed by one smaller (subdominant) cycle, and two much smaller (weak) cycles. Hypotheses on why cyclic dominance occurs include depensation due to overfishing (Collie and Walters 1987; Walters and Staley 1987; Walters and Woodey 1992); increased predation on the smaller subdominant or weak cycles (Ward and Larkin 1964; Larkin 1971); or alternatively, reduction in spawning abundance, juvenile rearing habitat, or food availability on these weak cycles, due to high spawner or juvenile Sockeye abundances on the dominant cycles. A review of hypotheses is presented in Levy & Wood (1992).

Cyclic fluctuations in abundance have changed over time for Fraser Sockeye populations (Cass and Wood 1994; Ricker 1997; Myers *et al.* 1998). From 1892 to 1912, most Fraser Sockeye populations cycled synchronously, with one dominant cycle line occurring every four years, followed by three weaker cycle lines (Figure 5). The dominant cycle occurred on the 1901 cycle line (for reference this would have been the 2009 cycle in current years), and appears to have persisted from as far back as the first reference to Fraser River Sockeye in 1793 (Fisheries and Oceans Canada 1998). During this early period of synchronous cyclic dominance (up to the 1913 Hells Gate landslide), the Fraser River was often considered to be the greatest Sockeye producer globally (Aro and Shepard 1967). Average returns from 1893 to 1913, on the 1901 dominant cycle, were 30.6 million Sockeye (Figure 5). Catch during dominant cycle years averaged 21.7 million and escapement averaged 8.9 million (Figure 5). On the remaining three weaker cycles, during this period, returns (average: 5.3 million annually), catch (average: 4.6 million annually) and escapement (average: 0.7 million) were considerably lower than the dominant (1901) cycle.

In 1913, construction work on the Canadian Pacific Railway line in the Fraser Canyon caused a major landslide at Hells Gate (see Figure 3 for location of Hells Gate in the Fraser Watershed). The landslide created an almost complete barrier to the large Fraser Sockeye populations that migrate into the upper watershed (Figures 3 & 5). The slide was particularly devastating because it occurred on the synchronous dominant cycle year for all Fraser Sockeye populations. As a result, the original dominant cycle (1901) was lost (Figure 5). After the 1913 landslide, considerable restoration work occurred at Hells Gate to permit upstream fish passage, and management actions were implemented to reduce overfishing and permit stocks to rebuild. Catch remained relatively high (2.1 million) despite the low Sockeye run sizes from 1914 to 1929 (average return: 2.4 million). After 1929, the run started to re-build slightly, with a less distinct dominant cycle occurring in 1930 (Figure 5).

Starting in the 1980's, the total Sockeye run built to a maximum return of 23.6 million (1993), and subsequently declined. During this recent period, highly cyclic stocks included the Shuswap Early Summer (Shuswap-ES) and Late (Shuswap Complex-L) runs (dominant cycle year: 2010), Quesnel (Quesnel-S), Early & Late Stuart (Takla-Trembleur-Stuart-S and Takla-Trembleur-EStu)(dominant cycle year: 2009) and Nadina (Nadina-Francois-ES) & Gates (Anderson-Seton-ES) (dominant cycle year: 2008). In recent years, returns have been particularly small (from 2007 to 2009 average return: 1.3 million; average escapement: 0.9 million; average catch: 0.4 million). In 2009, extremely low returns corresponded with the lowest productivity on record for most Fraser Sockeye stocks (Grant *et al.* 2010). In contrast, preliminary returns in 2010 were relatively high (~30 million preliminary returns as of October 11, 2010), and corresponded with average productivity for most stocks (above average productivity for Shuswap stocks). The mechanisms that produced the anomalously low returns of 2009 and the extremely high returns of 2010 remain uncertain, and are the subject of on-going scientific investigation (e.g. Peterman *et al.* 2010).

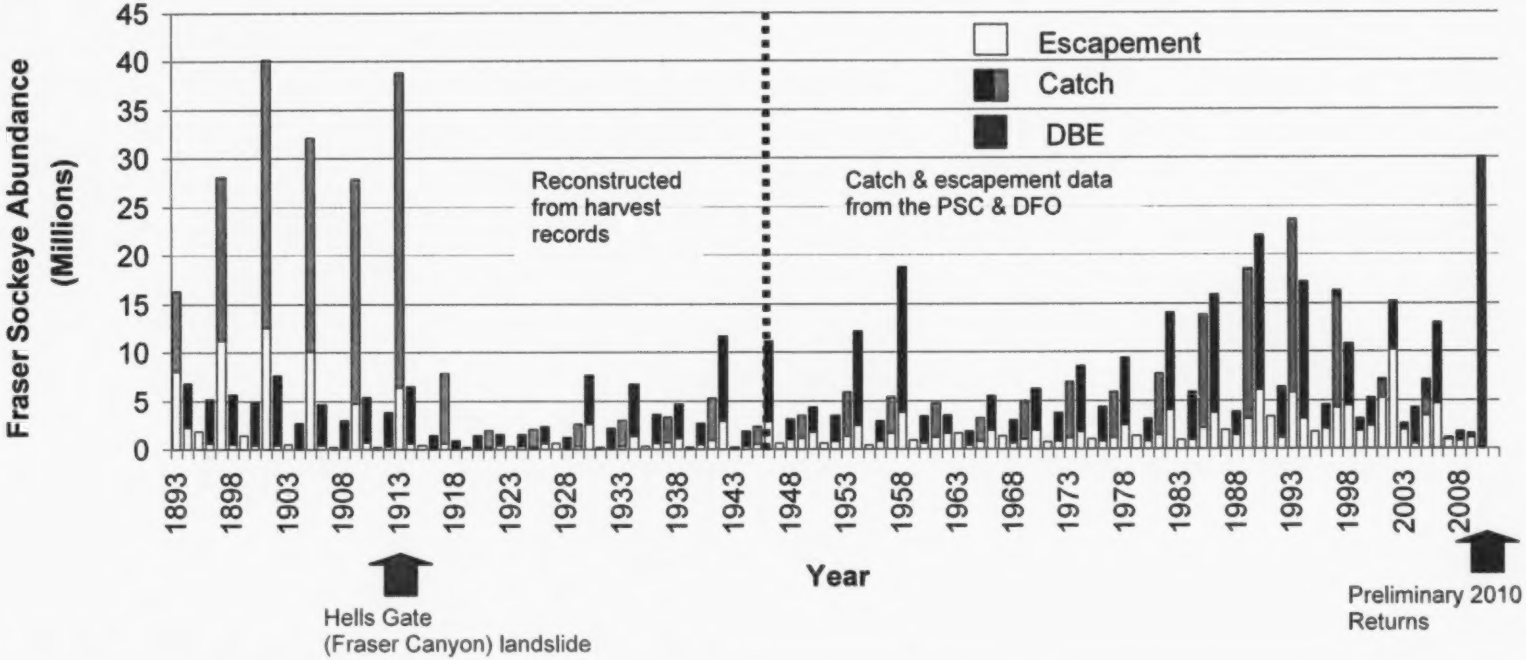

Figure 5. Escapement (white bars), catch (black or grey bars), and difference between estimates (DBE: red bars) for Fraser Sockeye. The 1901 (corresponding to 2009 in recent years) dominant cycle is represented by the grey bars. Maximum height of each annual bar represents the total return. The DBE is the difference between the in-season hydroacoustic estimates at Mission, BC and the spawning ground escapements plus in-river catches, and represents en-route loss and assessment errors. The dashed vertical line represents the division between early time series data (1982-1944), constructed by Gilhousen (1992) from commercial harvest data, and post-1944 data from the PSC and DFO records. Preliminary 2010 returns are indicated by the blue bar at the end of the time series. These data were provided by I. Guthrie & M. Lapointe from the PSC.

Productivity and Survival

Overall productivity (recruits-per-effective total spawner) for Fraser Sockeye was generally high up to the mid-1980's, and has subsequently declined (Figure 6 A). In recent years, productivity for some CUs has been below replacement (Appendix 3, Figure 1 d). The overall productivity trend in Figure 6A is driven by the most abundant CUs, which are largely Summer Run CUs (Chilko-ES/Chilko-S, Quesnel-S, Francois-Fraser-S (Stellako) and Takla-Trembleur-Stuart-S (Late Stuart)) that coincide with increases in escapement (Figure 6B). Amongst the remaining individual CUs, however, there is considerable variability in productivity trends (Grant et al. 2010; Appendix 3, Figures 1 c & d).

For the 17 current CUs and 2 *de novo* 'CUs' with stock-recruitment data, most CUs have exhibited a general decreasing trend in productivity, however, the timing of when this trend began differs amongst stocks (Grant et al. 2010; Appendix 3, Figures 1 c & d). Seven CUs have exhibited decreasing trends since the 1960's-1970's (Takla-Trembleur-EStu, Bowron-ES, North Barriere-ES (*de novo)*, Anderson-Seton-ES, Nadina-Francois-ES, Shuswap-ES and Seton-L). Eight CUs, including the four Summer Run CUs, have experienced decreasing trends starting in the 1980's-1990's (Cultus-L, Pitt-ES, Chilko-ES/Chilko-S, Takla-Trembleur-Stuart-S, Quesnel-S, Francois-Fraser-S and Lillooet-Harrison-L). Kamloops-ES (comprised of Raft River Sockeye), Shuswap Complex-L, and Harrison (U/S)-L (Weaver Creek & Channel) have not exhibited any persistent trends, and Harrison River (River-Type) has exhibited an increasing trend (with the exception of the 2005 brood year that represented the lowest productivity on record for Harrison) (Grant et al. 2010; Appendix 3, Figures 1 c & d).

To understand which broad ecosystem is driving changes in CU productivity, total survival can be partitioned into freshwater and marine survival, when both outmigrating smolt and adult return data are available. For Fraser Sockeye, only Chilko-ES/Chilko-S and Cultus-L Sockeye CUs have both smolt and adult return data. Most mortality in Fraser Sockeye occurs in the freshwater environment between the egg to smolt stage. Freshwater survival (smolts/egg), as indicated by Chilko River Sockeye, has been 3% on average, which is one third the average marine survival (adult recruits/smolt) of 9%. It is important to also note that marine survival estimates generally include some freshwater mortality, encountered in the Fraser River between the time smolts are counted exiting their rearing lakes and when they enter the marine environment. Chilko-ES/Chilko-S freshwater production has been exceptional in recent years; numbers of outmigrating smolts in the 2005 (77 million age-1 smolts) and 2006 (71 million age-1 smolts) brood years were well above average (1954-2009 brood years: 20 million age-1 smolts) for this CU (Figure 7 A). In contrast, both Chilko ES/Chilko-S and Cultus-L have experienced particularly low marine survival (below their cycle average) in the past four to eight brood years (Figure 7 B).

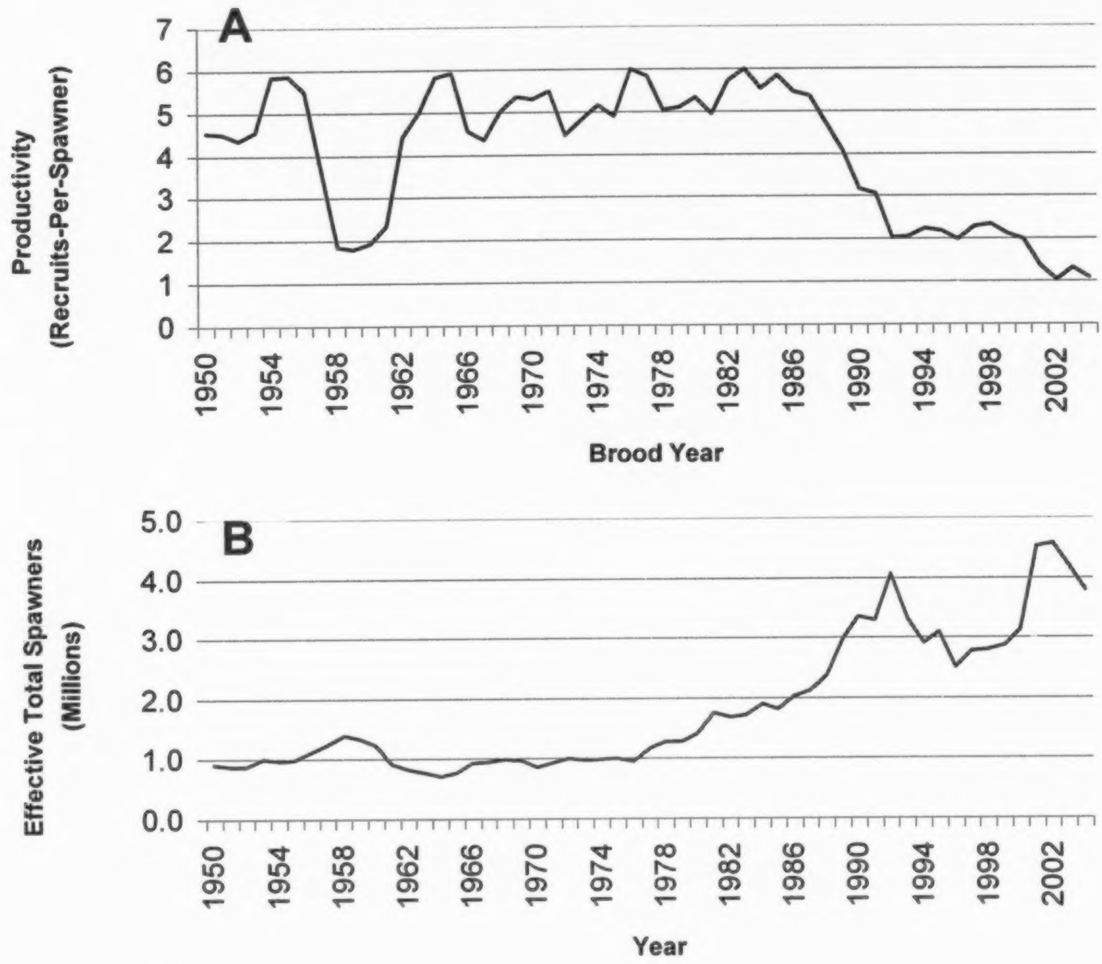

Figure 6. A) Four-year running average productivity in recruits (age-42 plus age 52)-per-effective-total-spawner, and B) escapement (effective total spawners) for Fraser Sockeye CUs. These trends are driven by CUs that dominate total abundance (Quesnel-S, Chilko-ES/Chilko-S, Takla-Trembleur-Stuart S, and Francois-Fraser-S). There is considerable variability in productivity trends amongst Fraser Sockeye CUs that deviate from this broad trend (Grant et al. 2010).

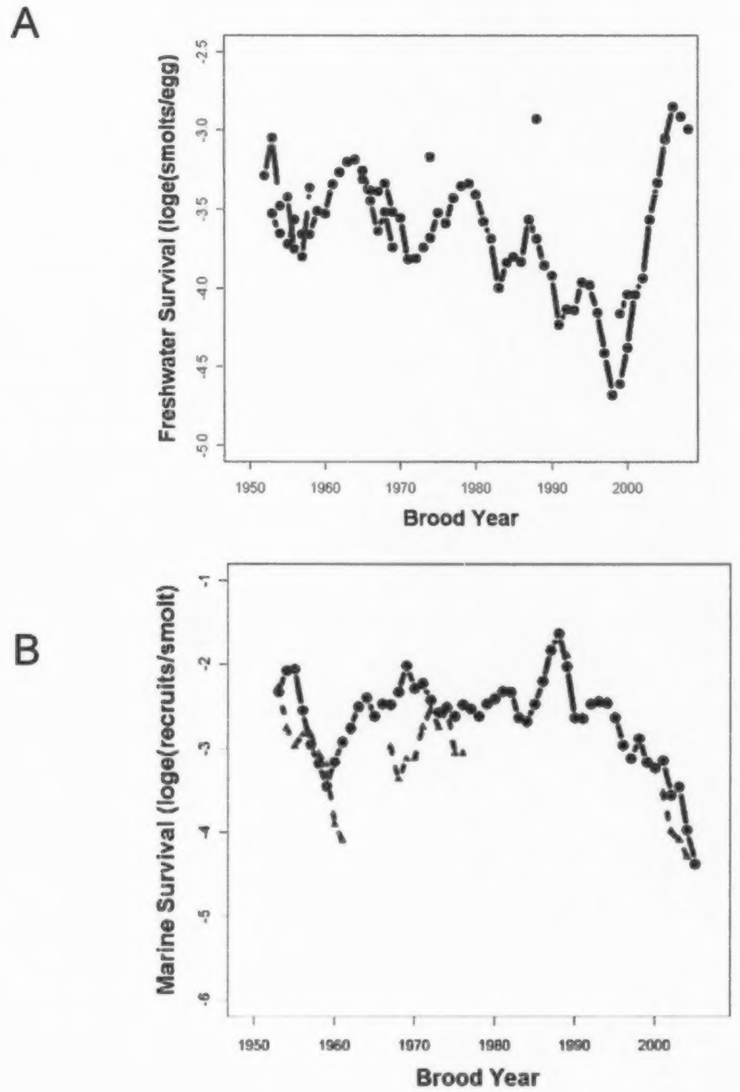

Figure 7. A) Chilko-ES/Chilko-S (blue solid line with circles) and Cultus (red solid line with circles) freshwater survival estimated as log$_e$ smolts/egg (eggs: number of effective female spawners multiplied by the average fecundity of 3,000 eggs/female). B) Chilko-ES/Chilko-S (blue solid line with circles) & Cultus-L (red dashed line with triangles) marine survival (log$_e$ recruits/smolt) from the 1951-2005 brood years. Note: the 2004 and 2005 brood year marine survival data include preliminary 2009 and 2010 age-4 and age-5 return data (these years are current in the process of being finalized). Cultus-L freshwater production in recent years (2000-2005) includes wild production only. Re-printed from Grant et al. (2010).

FRASER SOCKEYE POPULATION STRUCTURE

Genetics

The last glacial period is likely a major factor in the overall structure of current Fraser Sockeye populations. Two major glacial refugia influenced the current Fraser Sockeye population structure: the Cascadia refugia and the Beringia refugia (Wood *et al.* 1994; Withler *et al.* 2000; Beacham *et al.* 2005). Post-glaciation, the lower Fraser River was likely colonized by Sockeye moving in from the coastal North-Eastern Pacific (Beringia refugia), while the upper Fraser was likely colonized by Sockeye from the Columbia and Skeena Rivers (Cascadia refugia)(Wood et al. 1994). Recent genetic evidence has confirmed these two lineages for Fraser Sockeye (Beacham et al. 2005).

Since this last glacial period, the next known major event to affect particularly upper Fraser populations was the 1913 Hells Gate landslide, which blocked Fraser Sockeye passage and dramatically reduced abundances in the upper Fraser populations (see Figures 3 & 5). This event, in combination with relatively large fisheries, nearly extirpated Upper Fraser populations (Ricker 1950). Post-1913, despite both reduced population sizes and hatchery enhancement work in the Upper Fraser, there is little evidence that genetic bottlenecks (lower genetic variation) have occurred in Upper Fraser populations. Generally, transplants contributed little to the genetic variation of Fraser populations, with the exception of the upper Adams River (not a current CU since this new population is not persistent), Fennell Creek (North Barriere-ES) and Portage Creek (Seton-L) Sockeye (Withler *et al.* 2000). The original populations for these three CUs are considered extirpated (respectively, Adams-ES, North Barriere-ES and Seton-S; note: Seton timing changed from summer to late, after hatchery enhancement). The main factor contributing to the current genetic structure of Fraser populations is post-glacial colonization and limited straying from nursery lakes (Withler et al. 2000).

Run-Timing

The run timing groups of Fraser Sockeye were established for fishery management purposes, and, therefore, consist of populations with similar migratory timing during their return from the ocean to their spawning grounds. The earliest timed run is the Early Stuart Run, which is comprised of one Fraser Sockeye stock (Early Stuart: Takla-Trembleur-EStu CU) that spawns in the Takla-Trembleur watershed, and arrives in the lower Fraser River from late-June to late-July. The Early Summer Run, comprised of eight key stocks (Bowron: Bowron-ES; Raft: Kamloops-ES; Fennell: North Barriere (*de novo*); Gates: Anderson-Seton-ES; Nadina: Nadina-Francois-ES; Pitt: Pitt-ES; and Scotch & Seymour: Shuswap-ES) and a number of smaller stocks rolled up into an early summer miscellaneous group, spawn throughout the Fraser system, and arrive in the river from mid-July to mid-August. The Summer Run consists of four stocks (Chilko: Chilko ES/Chilko-S; Late Stuart: Takla-Trembleur-Stuart-S; Quesnel: Quesnel-S; and Stellako: Francois-Fraser-S) that arrive in the river from mid-July to early-September. The last run timing group to enter the Fraser watershed is the Late Run, which is comprised of six key stocks (Cultus: Cultus-L; Harrison: Harrison River (Rivery-Type); Late Shuswap: Shuswap Complex-L; Portage: Seton-L; Weaver: Harrison (U/S)-L; Birkenhead: Harrison-Lillooet-L) and a number of smaller stocks rolled up into a miscellaneous Late Run group, all of which enter the Fraser from late-July to mid-October. Historically, Late Run Sockeye held in the Strait of Georgia for three to six weeks prior to their upstream migration to their spawning grounds. However, starting in 1995, Late Run Sockeye migrated into the Fraser River with little delay in the Strait of Georgia, and this early entry was associated with high mortality rates. From 2002-2009, the Birkenhead population (Lillooet-Harrison CU) was separated from the Late Run group because their timing was more similar to Summer Run stocks, and because they did not exhibit high pre-spawn mortality similar to other Late Run stocks. However, starting in 2010, Birkenhead sockeye were re-integrated into the Late Run group because their timing had shifted to later than most Late Run stocks. The Summer-Run timing group typically dominates return abundances (Fisheries and Oceans Canada 2006; Fisheries and Oceans Canada 2008; Fisheries and Oceans Canada 2009), with the exception of the 2006 cycle, which is the dominant cycle for the Adams River Sockeye run (Shuswap Complex-L CU) (Grant *et al.* 2010). Overall, there is considerable overlap amongst all run timing groups.

Conservation Units

Methodology for the identification of conservation units (CUs) for Canada's salmon stocks in British Columbia (DFO's Pacific Region) is detailed in Holtby & Ciruna (2007). Also presented in Holtby & Ciruna (2007) was the original Fraser Sockeye CU list, subject to change as new data and information becomes available. This Fraser Sockeye CU list was recently modified (August 17, 2011)(Tables 1 & 2), after further consideration of Fraser Sockeye biology and escapement data. A number of CUs, and their associated escapement enumeration sites, changed in the latest Fraser Sockeye CU list (Tables 1 & 2; Holtby, DFO, pers. comm.). Changes to the current and extirpated CUs, removal of CUs from the original list, and the addition of CUs to the new '*de novo'* category, are documented in the proceeding CU results sections.

The first step in the identification of CUs for Fraser Sockeye, described by Holtby and Ciruna (2007), was to partition populations into two major life-history types: lake-type (populations that rear in freshwater as juveniles for one to three years) and river-type (populations that migrate to the ocean after gravel emergence). Subsequently, run-timing (for lake-type Sockeye), genetics, and freshwater-marine joint adaptive zones (for river-type Sockeye) were further used to identify and name individual CUs. Lake-type CUs include Sockeye populations observed in or above a lake or at a lake outlet (lakes larger than ~0.5 km^2) where there were no barriers to juvenile Sockeye passage. Generally, a single rearing lake that meets these conditions is used in the first part of a CU's name. There are cases where clusters of hydrologically connected lakes (<1 km^2) are combined into a single CU (e.g. Shuswap Complex-L), unless evidence exists to indicate these populations are genetically or ecologically distinct. Run timing (Early Stuart: EStu, Early Summer: ES, Summer: S; Late: L) is another factor used to distinguish between lake-type Sockeye CUs and, after the juvenile rearing lake, is used in a lake-type CU's name. Where data are available, lake-type Sockeye CUs are further partitioned into upstream (e.g. Weaver Creek and Channel populations that migrate upstream as fry to rear in Harrison Lake: Harrison (U/S)-L) and downstream lake migrants (e.g. the Big Silver population that migrates downstream as fry to rear in Harrison Lake as juveniles: Harrison (D/S)-L)(Holtby and Ciruna 2007). Conservation units were identified as new (*de novo)* if they were previously extirpated and, subsequently, were re-established through hatchery transplants from other systems (e.g. North Barriere-ES (*de novo)* and Seton-L (*de novo*)). Technically these hatchery origin populations do not meet the strict WSP definition of a CU, but these CUs are included in analyses as they contribute some production to overall Fraser Sockeye abundances.

River-type CUs do not meet the criteria outlined above for lake-type CUs, as these Sockeye do not rear in lakes after emergence from gravel. Instead, river-type Sockeye migrate to the ocean after gravel emergence. Although river-type CUs are typically named after the freshwater adaptive zone they occupy during spawning, there are a few Fraser Sockeye river-type CUs that are named after their spawning system. For example, river-type Sockeye that spawn in the Widgeon Slough and migrate to the ocean after they emerge from the gravel are named Widgeon (River-Type) and those that spawn in Harrison River and migrate to the ocean after they emerge from the gravel are named Harrison (River-Type) (Holtby and Ciruna 2007).

THREATS

The number of salmon that return to the Fraser River in any given year is determined by the number of effective spawners in the parental generation (specifically the number of eggs deposited in spawning gravel), age of maturity at return, and survival from the egg stage through to adult returns. Considerable mortality occurs in the freshwater and marine environments during the egg stage (egg incubation in lake or stream gravel), fry stage (lake rearing), smolt, and juvenile stages (downstream migration in the Fraser, Strait of Georgia ocean entry, and rapid northward migration through the Johnstone Strait, along the continental shelf to the North Pacific). Mortality can also occur in the adult stage prior to spawning, either en-route to the spawning grounds in the Fraser River, or on the spawning grounds (pre-spawn mortality). In addition, direct removal of adults through fisheries reduces the number of fish that reach their natal streams, rivers and lakes to spawn. A number of threats to salmon stocks in general, and to Fraser Sockeye populations in particular, have been identified and include the following: fisheries, environmental conditions in the freshwater and marine environments, en-route and pre-spawn mortality, habitat alteration, exotic species, and pathogens and disease.

Fisheries

Management: past & present

From 1946-1984 the International Pacific Salmon Fishery Commission (IPSFC) was responsible for management of Canadian (British Columbia) and United States (Washington State) fisheries in an area known as the Convention Area. The total allowable catch (TAC) of salmon in this area was shared equally between Canada and the United States. Since 1985, following the Pacific Salmon Treaty, the Fraser River Panel (FRP) of the Pacific Salmon Commission (PSC) has regulated management of Fraser Sockeye fisheries in Panel Area waters (updated January 27, 2009: www.psc.org/pubs/Treaty.pdf (Figure 8). The Fraser River Panel is comprised of Canadian and U.S. representatives, and its purpose is to ensure that spawning escapement targets for each major stock or stock group, set by Canada as well as international and domestic allocation goals, are met (Fisheries and Oceans Canada 1998). Under the Treaty, the U.S. share of the Fraser Sockeye harvest has gradually decreased; under the current annex it is 16.5% of international TAC. DFO is responsible for management of the Canadian fisheries outside the Panel area (Figure 8), but must coordinate actions with the Fraser Panel (FRP) to ensure that escapement and allocation objectives are met. Annually, DFO produces a Southern B.C. Salmon Integrated Fisheries Management Plan (IFMP) for all salmon fisheries in BC waters, which incorporates the results of consultations, and input from First Nations, commercial and recreational sectors, and NGOs. The IFMP provides specific decision rules for a number of salmon fisheries, including those directed at Fraser River Sockeye (see IFMP's on the following DFO Website: http://www-ops2.pac.dfo-mpo.gc.ca/xnet/content/MPlans/MPlans.htm.

Management of Fraser River Sockeye is highly complex, since there are approximately 19 major stock groups with inter-annual differences occurring in abundance and migration timing. Under the terms of the Treaty, fisheries are managed using information on four run-timing aggregates: Early Stuart, Early Summer, Summer and Late. Typically, several stocks will co-occur in the primary fishing areas because of similarities in the marine arrival and upstream migration timings of different stocks. In addition, the diversion rate (proportion of Fraser Sockeye stocks approaching the Fraser River via the northern route through Johnstone Strait versus the souther route through the Juan de Fuca Strait) varies considerably both within and between years. For these reasons, and because of the different escapement objectives for each stock, Fraser Sockeye management decisions frequently involve trade-offs between harvest and escapement objectives of the various stock groups. For example, it is not uncommon for some fraction of the harvest of more abundant stocks to be foregone to protect less abundant stocks with similar migration timing.

Fishing plans for Sockeye are based initially on pre-season forecasts of stock abundance, diversion rates (through Johnstone Strait versus Juan de Fuca Strait), and migration timing. Typically, contingency plans are developed on a range of forecast values, including potential forecasts that are both lower and higher than the median predictions. Pre-season plans are later refined by in-season estimates of return abundance, derived from relative abundance indices in test fisheries, estimates of lower river escapements from the PSC hydro-acoustic facility at Mission (B.C.), data from other sources of harvest, as well as stock composition analysis. Fishery openings and closures in Panel waters are managed by the FRP to achieve target escapement levels for the four run timing groups. Canada co-ordinates its Fraser Sockeye fisheries outside Panel waters to ensure they are consistent with international and domestic objectives. Both Canada and the U.S. adjust fisheries directed at Fraser Sockeye to minimize interceptions of non-target species such as Pink, Chum, Chinook, Coho and Steelhead salmon, and to limit catches of stocks of concern, such as interior Fraser Coho, Steelhead, Sakinaw and Cultus Sockeye. Throughout the fishing season (June to late September), estimates of Sockeye run size and stock composition are constantly revised, and management responds with adjustments to fisheries decisions (based on changes in run estimates, spawning escapement objectives, gross escapement objectives, and available TAC). Gross escapement objectives include the spawning escapement targets plus any in-river catch requirements, and an additional factor called a management adjustment. Management adjustments are additions to the spawning escapements targets that ensure that the number of fish reaching spawning areas will reach desired levels. Management adjustments account for both systematic differences between upper and lower river escapement estimates, as well as in-river migration conditions. River migration conditions are monitored daily, and management adjustments are updated frequently during the in-season period based on the combination of observed and forecasted river conditions.

Information on in-season changes are provided on the PSC website: www.psc.org/news_frpnews.htm and through DFO Fisheries Notices: http://www-ops2.pac.dfo-mpo.gc.ca/xnet/content/fns/index.cfm?pg=search_options&lang=en&id=recreational.
After each fishing season, FRP management decisions and strategies are assessed to determine if goals were met, and to look at options for improving management, data collection, and analysis techniques.

Catch History

The first cannery was built on the Fraser River in 1866, spurring rapid development of the commercial gillnet fishery. Relative to total returns, this fishery was particularly intense on the subdominant cycles (Figure 5). It is likely that fisheries exaggerated the cyclical pattern of return abundances, maintaining weak cycles (relative to the dominant 1901 cycle) due to depensatory exploitation rates. Prior to 1913, catches ranged from 1.8 to 32.3 million (Figure 5). After the Hells Gate landslide, upstream passage was greatly restricted for several years, and subsequent overfishing further constricted Sockeye abundances. As a result, catches declined to an average of 1.9 million fish from 1915-1930 on all cycles (Figure 5) (Fisheries and Oceans Canada 1998). Exploitation rates (catch/total return) were again high from 1950 to the mid-

1990s (average: 75%), and have subsequently declined (average: 34%). The highest catch since 1958 occurred in 1993 (17.8 million Fraser Sockeye caught), with 95% of this occurring in marine areas. In recent years (2007-2009), catches have been the lowest on the time series (average: 2.5 million), due to extremely low returns in these years. During these years the majority of harvests in Canada were allocated to meet First Nation's FSC (food, social and ceremonial) needs. However, on larger return years, most catch still occurs in the marine areas; the majority of the Canadian commercial harvest is caught in the troll fisheries, the purse seine and gillnet fisheries in Johnstone and Juan de Fuca Straits, and the gillnet fishery in the Fraser River. Additional commercial catch occurs in smaller commercial fisheries in northern and central B.C. that occasionally intercept Fraser River Sockeye, and in a few directed fisheries in the Strait of Georgia that primarily target Late Run Fraser stocks. Other Fraser Sockeye catch occurs in the native food fisheries that operate throughout the Fraser River watershed, and in the recreational fishery, which has increased its catches since the mid-1990's, with the development of the in-river fishery upstream of Mission. United States (U.S.) catches mainly occur in net fisheries in the southern approaches to the Fraser River, specifically the U.S. waters in the Juan de Fuca Strait, near the San Juan Islands, and south of Point Roberts. Some Fraser Sockeye are also taken incidentally in southeastern Alaska. United States catches are generally small, averaging at 18% of total catch since 1993.

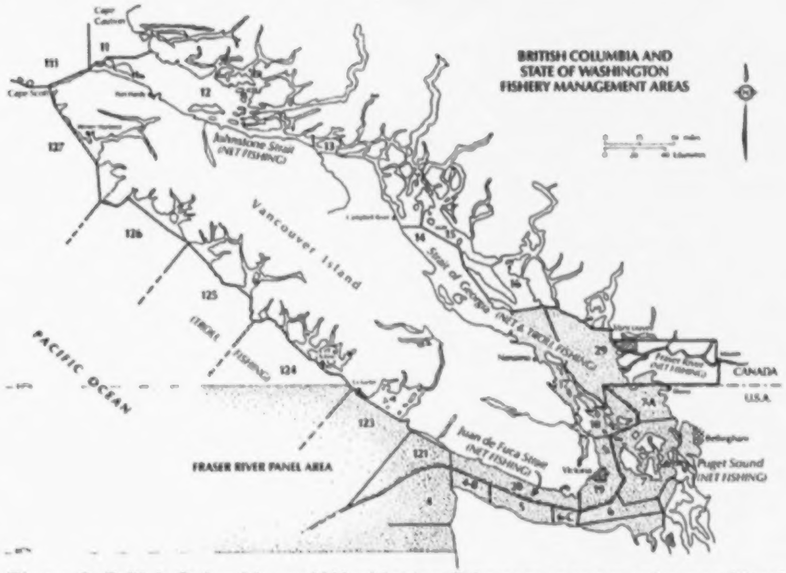
BRITISH COLUMBIA AND
STATE OF WASHINGTON
FISHERY MANAGEMENT AREAS
Cape Caution
Cape Scott
111
11
12
127
Johnstone Strait (NET FISHING)
13
126
Vancouver Island
14
15
Strait of Georgia (NET & TROLL FISHING)
16
125
(TROLL FISHING)
PACIFIC OCEAN
124
29
Fraser River (NET FISHING)
CANADA
U.S.A.
7A
123
121
Juan de Fuca Strait (NET FISHING)
Victoria
Bellingham
Puget Sound (NET FISHING)
FRASER RIVER PANEL AREA
4-B
20
5
6-C
6
7
4

Figure 8. British Columbia and Washington fishery management areas http://www.psc.org/ including Fraser River Panel Area waters (shaded grey).

Environmental Conditions (Freshwater Author: M. Hague, Science, DFO)

Freshwater Environment

Freshwater life history stages account for a significant proportion of overall mortality and variation in total mortality (>40%) in Sockeye salmon species (Bradford 1995). The transition and migration between habitats at critical life stages expose Sockeye salmon to high levels of mortality. Examples of high levels of mortality at critical stages include high mortality rates (60-90%) reported for egg to fry stages in Takla River (D. Patterson and M. Hague, DFO, pers. comm.), estimates of high mortality (>50%) during smolt outmigration for Cultus and Chilko Sockeye smolts (Welch et al. 2008)(S. Hinch, UBC, pers. comm.) and Fraser Sockeye premature mortality (often exceeding 20%) during freshwater migration and spawning (Gilhousen 1990; Peterman *et al.* 2010). The combined effects of mortality in different freshwater life history stages have ultimately been linked to overall changes in productivity and abundance of salmon populations. Rates of population decline and variability in total survival have sometimes been attributed, in part, to indices of freshwater habitat condition (Bradford and Irvine 2000; Mueter *et al.* 2005).

In contrast to the marine environment, specific freshwater processes controlling survival are generally well identified. During the egg development stage, survival has been directly linked to water quality issues, such as temperature, sedimentation, metals, and dissolved oxygen in the spawning environment (Levasseur *et al.* 2006; Greig *et al.* 2007). Scouring by high winter flows (Steen and Quinn 1999) or dewatering due to low water levels (Neitzel and Becker 1985) are also a concern. Estimates of productive capacity for Fraser Sockeye lakes have been forecasted from photosynthetic rate (PR) models (Shortreed et al. 2001), and can be used as an index of lake rearing suitability. Recent studies show that density-dependent growth rates of fry are also mediated by interactions with lake temperature (Crozier et al. 2010). There is no direct data linking smolt survival with environmental conditions for Fraser Sockeye, but other studies have shown positive relationships between spring flows and smolt outmigration survival (Kjelson and Brandes 1989). Adult spawning migration survival is a function of both acute and cumulative impacts largely mediated by exposure to extreme temperatures and flows (Gilhousen 1990; Wagner *et al.* 2005; Crossin *et al.* 2008; Macdonald *et al.* 2010; Mathes *et al.* 2010).

There are a multitude of factors influencing salmonid freshwater survival, but temperature indices are often used to summarise the overall quality of freshwater habitat (Nelitz et al. 2007) since many physiological and phenological processes are related to thermal conditions (Brett 1971). Furthermore, significant trends in warming freshwater temperatures (Quinn and Adams 1996; Foreman *et al.* 2001; Patterson and Hague 2007; Patterson *et al.* 2007a) and changes in hydrology (Rodenhuis *et al.* 2007; Pike *et al.* 2008) are consistent with changes in river entry timing and behaviour for salmon populations in the Columbia River system (Quinn *et al.* 1997; Goniea *et al.* 2006), as well as an increased frequency of high en-route loss events for Fraser River Sockeye salmon (Macdonald *et al.* 2010; M. Hague and D. Patterson, DFO, pers. comm.). Despite basin-wide temperature increases, and the role of temperature in mediating growth and survival at juvenile life stages, the limited data available shows no consistent trends across populations. Similarly, the increase in water temperatures and the expected changes in the timing of migration for fry (e.g. Stuart/Takla DFO data), smolts (Mission timing DFO data), and adults (Late Run) have not occurred.

There is a general consensus that the Fraser River will continue to warm throughout the 21st century and will likely shift from a predominantly snowmelt to a rainfall driven system (Morrison *et al.* 2002; Ferrari *et al.* 2007; Nelitz *et al.* 2009). These changes could alternatively be exacerbated (pine beetle, forest harvest, groundwater) or potentially mitigated by anthropogenic activities occurring during the same time period (Nelitz *et al.* 2009; McDaniels *et al.* 2010). Climate change has the potential to impact all salmon freshwater life history stages, however, experts have identified the freshwater egg-to-fry and adult spawning migration as being the most susceptible (McDaniels *et al.* 2010). If warming trends continue as anticipated, the majority of Fraser River Sockeye salmon populations are generally expected to suffer from increases in the frequency and magnitude of en-route loss events (Martins *et al.* 2011), and we may also anticipate basin-wide declines in egg and fry survival (McDaniels *et al.* 2010).

Marine Environment

In addition to freshwater conditions, ocean environmental conditions also contribute to both large interannual variations in productivity (recruits/spawner) of salmon as well as longer-term persistent changes in average productivity (Mantua *et al.* 1997; Beamish *et al.* 1997; Beamish *et al.* 1999; Beamish *et al.* 2004b). The mechanisms that link changes in climate to changes in salmon productivity are poorly understood. However, it is generally thought that salmon are most vulnerable in the first six months after ocean entry (early ocean entry to first over-winter period) when they are their smallest size and, therefore, most vulnerable to the two major mortality mechanisms, predation and starvation (Beamish and Mahnken 2001). In particular, it is hypothesized that during the early ocean entry period salmon are particularly vulnerable to predation due to their small size, and that during their first ocean over-winter period they are most vulnerable to starvation and that reaching a critical size is the key to over-winter survival (Beamish and Mahnken 2001). For Fraser Sockeye, given that almost all populations enter the Strait of Georgia as smolts and then generally rapidly migrate northward through the Johnstone Strait, along the continental shelf and out into the North Pacific (Tucker et al. 2009; Welch et al. 2009), there is a broad area over which these fish will be particularly vulnerable to early marine mortality.

Longer-term fluctuations in salmon abundance have been linked to broad changes in ocean climate that start with changes in major pressure systems over the Pacific, affecting ocean temperatures and productivity. Two key indices of the climate-ocean system include the Aleutian Low Pressure Index (ALPI) (Beamish *et al.* 1997) and the Pacific Decadal Oscillation (PDO) (Mantua *et al.* 1997). Positive ALPI (a measure of the intensity of the Aleutian Low pressure system in the North Pacific) indicate large Aleutian Lows and decreased upwelling along coastal North America; negative values indicate the opposite. Positive PDO (an index of sea surface temperatures in the Pacific) indicate warmer temperatures along the west coast of North America and cooling in the central Pacific; negative PDO's indicate the opposite. In summary, positive ALPI and negative PDO's represent improved ocean conditions for salmon. There has been evidence of major shifts in these indices in 1925, 1947, 1977, 1989 (ALPI only), and 1998 (Beamish *et al.* 1997; Beamish *et al.* 1999; Beamish *et al.* 2004a; Beamish *et al.* 2004b; Beamish *et al.* 2004c). Specifically, 1977 to 1988 was a productive period for Sockeye Salmon (Beamish *et al.* 2004b) followed by a period of decreased productivity in the 1990's. This coincides with a period of increasing numbers of returning Fraser Sockeye up to the mid-1990's and a subsequent decrease in abundance (Figure 5). In addition to broader changes in ocean conditions, regional-scale factors such as sea-surface-temperature have also been used to predict survival rates in salmon (Mueter *et al.* 2002; Mueter *et al.* 2005).

Despite these linkages to broad scale and regional climate patterns in the ocean, predicting future survival of Fraser Sockeye salmon remains a challenge (Haeseker *et al.* 2008; Grant *et al.* 2010). There is likely a complex set of conditions in both the freshwater and marine environment (temperature, food availability, and predation) covering a broad temporal and spatial scale, that determines survival and total recruitment for Fraser Sockeye stocks. These conditions likely vary interannually, and therefore, no one factor is sufficient to explain variability in Fraser Sockeye recruitment.

Early Migration and Pre-Spawn Mortality (Author: D. Patterson, Science, DFO)

Pre-Spawn Mortality

The historic pre-spawn mortality (PSM), quantified as population estimates of the percentage of egg retention in female carcasses recovered from the spawning grounds, for Fraser Sockeye salmon populations averages from 10 to 15% across populations, with extreme events (>40%) being episodic and highly variable among stocks. The causes and associations of PSM are complex and multi-factorial (Gilhousen 1990) and include pathogens, high stress and low energy, and longevity on spawning grounds (Macdonald *et al.* 2000; Macdonald *et al.* 2007; Crossin *et al.* 2008; Bradford *et al.* 2010b; Bradford *et al.* 2010c). Again, most of these factors are accentuated by increasing temperatures and increased time spent in freshwater. Therefore, it is not surprising that within-stock trends in PSM are correlated with migration timing and/or migration and spawning ground temperatures (Gilhousen 1990). Correlations with temperature also increase with proximity to spawning ground (Macdonald *et al.* 2007). While there are no consistent trends across stocks, there is some evidence that PSM has been higher and more variable in recent years for Late Run stocks (Hinch 2009), and in 2008 a system-wide PSM event resulted in overall poor egg retention of 64% (DFO data).

Late Run Sockeye Early Migration

An extreme example of a threat to Sockeye salmon from both en-route and pre-spawn mortality comes from a closer examination of Late Run Sockeye populations over the past 16 years. During this time period, Late Run Sockeye have on average entered the Fraser River approximately three to six weeks earlier than normal without a change in spawning dates (Lapointe *et al.* 2003; Cooke *et al.* 2004). The early entry is a result of a reduced holding period in the Strait of Georgia, as marine approach times have not changed. This has resulted in Late Run Sockeye being exposed to higher en-route migration temperatures, associated with late August/early September arrivals, for longer periods of time. The combination has contributed to high en-route loss estimates (especially for the early entrants (English *et al.* 2005)) and high PSM values in recent years. While the causes for the shift in early entry behaviour have proved elusive (Hinch 2009), the consequences have been well documented. In 2009 and 2010, however, Late-run Sockeye reverted to more normal entry timing.

Habitat Alteration

Fraser Sockeye have specific habitat requirements during their freshwater life-history stages (including their entry into freshwater as adults and subsequent upstream migration to spawning grounds; egg incubation in lake or river gravel; juvenile lake rearing; and downstream outmigration as smolts enroute to the Pacific Ocean). Given significant mortality occurs in the freshwater environment, habitat alteration in the freshwater may impact total survival and, therefore, total recruitment for Fraser Sockeye. Throughout the watershed, urban development, transportation corridors, agricultural and forestry land-use, recreational land and water-use, water extraction, etc. represent risks to Fraser Sockeye during their freshwater residence.

Although water quality issues have not been identified as a watershed-wide concern for Fraser Sockeye, there are localized water quality issues that represent risk to Fraser Sockeye survival in the freshwater environment. In particular, all Fraser Sockeye populations must migrate through the highly urbanized Lower Fraser River area during both their upstream migration as adults and their downstream outmigration as juveniles. The greatest concentration of human development within the Fraser watershed occurs in the Lower Mainland near the outlet of the Fraser River (83% of total development in the Fraser watershed) (Schreier *et al.* 1991). In this area, Fraser Sockeye may be exposed to contaminant inputs from point sources (e.g. waste-water treatment plants) and non-point sources (e.g. urban run-off) that can result in fish mortality or may interfere with migration timing, homing behaviours, and physiological transitions into the marine or freshwater environment. Sources and contaminants in the Strait of Georgia are presented in detail in Grant and Ross (2002) and those in the Fraser watershed are found in Johannessen and Ross (2002). Details on specific risks and impacts to Fraser Sockeye are also documented (Johannessen and Ross 2002). Other localized impacts also occur, particularly in lake environments with foreshore human development such as Cultus Lake and Shuswap Lake (Main Arm), where agricultural runoff, foreshore septic systems, houseboats and other lake recreation can input deleterious substances into the lake environment. Recent studies in Shuswap Lake, for example, have detected notable declines in water quality.

Gravel removal for flood control, which has occurred in recent years in the Lower Fraser River (downstream of the Fraser Canyon) between Hope and Mission, has been flagged by stakeholders as a concern to Fraser Sockeye. However, currently there is no indication that gravel removal impacts Fraser Sockeye during their upstream migration as adults or downstream migration as smolts.

In the Upper Watershed (above the Fraser Canyon), forestry is the single largest land use activity. Observed land-use issues related to forestry have included stream crossings impairing fish migration, sediment input, riparian vegetation impacts etc. Generally, however, habitat issues related to forestry have not been regarded as significant issues to Fraser Sockeye. More recently, the Mountain Pine Beatle (MPB) (*Dendroctonus ponderosae)* has represented a major issue related to forests and forestry in the upper watershed as it expands its range due to milder winters. The MPB has affected a significant portion of the Fraser watershed by killing huge areas of forest. The change in forest coverage due to both the MPB killed trees and resultant salvage logging is predicted to cause significant hydrological changes in the watershed, changing the nature and timing of peak flows, low flows and temperature regimes, and has the potential to change riparian communities and sedimentation.

Water use and withdrawal for human use, occurring particularly in the Upper Fraser Watershed, has been identified as a concern for certain waters that support Sockeye. Due to increasing demands for water, reduced supply due to climactic variability, and the existence of long-standing historical water rights, the availability of water for fish may be significantly reduced in the near future. As an example, in southern and interior BC the natural period of low water levels (flow) during the summer often coincides with peak irrigation demand, as well as the migration and spawning period for salmon. A combination of these factors can significantly impair the ability of salmon to successfully migrate and spawn, as has been observed in the Thompson-Shuswap and Chilcotin areas.

Exotic Species

Exotic (non-native) fish species represent potential threats to salmonid populations in British Columbia (Tovey *et al.* 2008; Bradford *et al.* 2008a; Bradford *et al.* 2008b). Non-native fish species have largely expanded their distribution outside of their natural ranges through stocking programs that occurred as early as the 1800's (Rahel 2002). Due to the recognition of the risks to native biota and ecosystems, stocking of non-native fish species has been more conservative in the last two decades (Rahel 2002). However, non-native species continue to be introduced into aquatic ecosystems through both unauthorized introductions by the public or through continued expansions of their ranges beyond their initial point of introduction. Six exotic fish species, in particular, present a risk to Fraser Sockeye, and include the Yellow Perch (*Perca flavescens*), Pumpkinseed (*Lepomis gibbosus*), Northern Pike (*Esox lucius*), Walleye (*Sander vitreus*), Smallmouth bass (*Micropterus dolomieu*), and Largemouth bass (*Micropterus salmoides*) (Tovey *et al.* 2008; Bradford *et al.* 2008a; Bradford *et al.* 2008b). For Perch, Smallmouth and Largemouth Bass, the probability of becoming widely established once it has arrived in BC is considered high (Bradford *et al.* 2008a; Bradford *et al.* 2008b). Other species such as Pike, Walleye, Pumpkinseed present high risks to native biota if they spread further in BC (Tovey *et al.* 2008). Depending on the invasive fish species, they can either compete for food resources (i.e. Perch and Pumpkinseed) or are predators (i.e. Pike, Walleye, Smallmouth and Largemouth Bass) of juvenile Fraser Sockeye in their rearing lakes.

Pathogens and Disease (Author: K. Garver, Science, DFO)

A diverse range of pathogens including viruses, bacteria, fungi and parasites can infect Sockeye Salmon. However, it is important to note that the presence of a pathogen in a Sockeye Salmon does not necessarily result in disease or compromised heath conditions. Whether or not a Sockeye Salmon becomes diseased when exposed to a pathogen depends upon complex interactions between the host, the pathogen and the environment in which these interactions take place. Disease can present itself in Fraser Sockeye Salmon lethally or sublethally (e.g. changes in swimming ability, growth, osmocompetence and reproduction). However, quantification of these disease impacts in wild fish can be difficult. Due to the overall complexity of disease it is extremely difficult to predict the occurrence and severity of disease and what, if any, role disease plays in structuring Fraser River Sockeye populations.

Three pathogens that have been directly observed in Fraser Sockeye include infectious hematopoietic necrosis (IHNN) virus, *Ichthyophthirius mutifiliis* and *Parvicapsula minibircornis*. Infectious hematopoietic necrosis virus (IHNV) is an aquatic rhabdovirus that is enzootic (constantly present) in Sockeye salmon populations in the Pacific Northwest of North America. The virus infects all life history stages of Sockeye salmon, however IHN disease is predominantly observed in fry, while adult spawning Sockeye, although carriers of virus, remain asymptomatic. Mass mortality events due to IHNV disease have been reported in two Fraser River Sockeye stocks. The first IHNV mortality event occurred in the spring of 1973 at Chilko Lake, and resulted in an estimated loss of 23.7 million fry. Subsequently, in 1987 an IHNV epizootic event occurred at Weaver Creek spawning channel resulting in nearly 50% mortality (8.3 million fry died out of a total 16.8 million) of all migrating fry within days of leaving the spawning channel. Despite these significant impacts incurred in Fraser Sockeye fry due to IHN disease, long-term monitoring of Nadina River and Weaver Creek spawning channels has revealed that over a 24-year period (1986-2009), IHNV prevalence varies annually within the same Sockeye stock and is inconsistent between stocks. There is no correlation with IHNV prevalence in adults and the occurrence in fry. Additionally, the data set illustrates that the occurrence of IHN disease outbreaks in fry have not increased over the 24 year monitoring period for either Weaver Creek or Nadina River stocks. Our inability to detect IHNV in Sockeye salmon fry from Weaver Creek and Nadina River over the past 10 (1998-2007) and 16 (1992-2007) years; respectively, suggests that IHNV is not a major contributor to the long-term decline of these two stocks.

Ichthyophthirius multifiliis (ICH) is a naturally occurring freshwater ciliate protozoan that causes a disease commonly referred to as "ich" or "white spot disease". The pathogen typically does not cause disease in Sockeye salmon. However, if conditions such as warm water, reduced flows, and adult crowding exist then disease can occur due the development of high numbers of this pathogen. Such disease events have been documented in Fraser and Skeena River Sockeye salmon and have resulted in severe pre-spawn mortalities of up to 80%. However, as with IHN disease, ICH disease prevalence has been inconsistent and varies between stocks. Additionally, the frequency of ICH epizootic disease events at Weaver Creek and Nadina River has not increased since 1990, suggesting that ICH disease is not a major factor contributing to the long term decline of these two stocks.

Parvicapsula minibicornis is a myxozoan parasite that is enzootic in Fraser River Sockeye stocks. Surveys for the parasite have revealed that transmission occurs at or near the river estuary and that adults and juvenile salmon become infected with the parasite as they migrate through this area. In adult salmon, the prevalence and severity of infection is affected by time and temperature, such that migrating Sockeye holding in the river under elevated river temperatures are at higher risk of more severe infections. Severe *P. minibicornis* infections may interfere with renal osmoregulatory function and increase the probability of pre-spawning mortality. However, assigning a clear negative impact due to this parasite is difficult, as severe *P. minibicornis* infections are also evident in successfully spawning fish. There are no data on the severity of infection of juvenile Sockeye in marine waters with *Parvicapsula*. In the absence of information regarding the relationship between *Parvicapsula* infection and disease in Sockeye salmon, its contribution to migratory behaviour and/or high mortality remains unknown. In summary, pathogens are a natural component of all ecosystems and not all infections lead to disease. Often enzootic pathogens are 'well-adapted' in that they do little to harm their host, however, the incidence and severity of disease from such pathogens may increase if abnormal conditions and/or adverse factors ("stressors") occur.

STATUS ASSESSMENT METHODS

DATA

Escapement Data

In the early 1900's, spawner abundance was estimated by the Government of Canada's Fisheries Agency using visual techniques that were often opportunistic and not specifically designed for the systems being assessed. In 1938, additional resources became available for the development of improved estimation techniques, and concurrently the International Pacific Salmon Fisheries Commission (IPSFC) assumed responsibility for the management and assessment of Fraser River Sockeye resources. The IPSFC's early work (Atkinson 1944; Howard 1948; Schaefer 1951) resulted in a two-tiered escapement approach, with higher precision assessment methods applied to stocks that were predicted to return at higher abundances and lower precision methods applied to stocks predicted to return at lower abundances (Woodey 1984; Andrew and Webb 1987).

With the signing of the Pacific Salmon Treaty in 1985, Fisheries and Oceans Canada (DFO) assumed responsibility from the IPSFC for the assessment of Fraser River Sockeye, and adopted the two-tiered escapement estimation system developed by the IPSFC, whereby the method of estimation for each CU was based on the number of spawners expected to return in a given year. Historically, low precision visual surveys have been used to enumerate stocks with expected low escapements (<25,000 spawners). For stocks with large expected returns (>25,000), higher precision methods, such as enumeration fences and mark-recapture programs, were used. In 2004, this threshold was raised to >75,000 spawners to reduce the number of stocks assessed with high precision methods due to funding limitations. Calibration work by DFO is on-going to assess visual survey expansion factors for these larger population sizes. Starting in the mid-1990's, the number of assessed sites increased across a number of larger Fraser Sockeye CUs, due to improvements in equipment (e.g. boats) and funding that permitted increased spatial assessment coverage of smaller Sockeye spawning streams.

Escapement enumeration methods for Fraser River Sockeye salmon are documented in a number of technical reports (Houtman and Cone 1995; Schubert and Tadey 1997; Schubert and Fanos 1997a; Schubert and Fanos 1997b; Schubert 1998; Cone 1999; Houtman *et al.* 2000; Schubert 2000; Schubert 2007; Schubert and Houtman 2007). Annual escapement plans are also available on-line: http://www.pac.dfo-mpo.gc.ca/fraserriver/escapeupdate.htm. Fence and tower counts are considered the most accurate methods of estimating spawner abundance, with almost all fish being counted as they migrate past, barring operational or environmental constraints. Fence counts are typically used to calibrate less accurate visual surveys and to estimate bias in mark recapture programs. Visual surveys are conducted by air (helicopter) or ground (boat or foot) and are considered the least accurate and precise methods to assess salmon abundance. Visual counts are expanded to total escapement based on calibration work, in which fence counts were conducted simultaneously with visual surveys on smaller creeks with generally good visibility. Although a factor of 1.8 is applied to expand escapement counts from visual surveys to estimate total escapement (Andrew and Webb 1987), recent calibration results indicate that this factor typically underestimates actual escapement (estimates are negatively biased) in larger systems (both size of system and numbers of spawners) (K. Benner, DFO, pers. comm.). Mark recapture estimates fall somewhere between fence/tower counts and visual surveys for accuracy and precision. Bias in mark recaptures is generally identified and corrected for in the analyses.

Escapement data (total number of adults that 'escaped' fisheries and were enumerated on the spawning grounds) are recommended by Holt et al. (2009) to evaluate trends in abundance for Pacific Salmon. For Fraser Sockeye however, additional data on the spawning success of female fish are also available, and are therefore used in the current paper to estimate status for *trends in abundance* and *abundance* metrics. Spawner success for a population is calculated as the proportion of eggs (0%, 50%, or 100%) successfully spawned, based on spawning ground carcass surveys. For *trends in abundance* metrics, effective female spawner (EFS) data are used (product of the number of female spawners and spawner success) as egg production is limited specifically by the number of females that have successfully spawned. For *abundance* metrics, effective total spawner (ETS) data are used (product of the total adult male plus female spawners and female spawner success). Both males and females (instead of just females in the case of the *trends in abundance* metrics) are used for *abundance* metrics since total abundance benchmarks are more readily transferable to fisheries management applications and to other status assessments such as those conducted by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC). Effective total spawners data, however, similar to effective female spawners, are calculated using female spawner success in its calculation to more closely reflect successful spawners.

For most CU data, the start of the escapement (EFS & ETS) time series was truncated to 1950, since earlier assessments were often conducted opportunistically using visual survey methods not specifically designed for the system being assessed. There are some CUs for which the escapement time series starts later than 1950, and these are documented in the proceeding individual CU sections. At the time of this report, the most recent escapement data available was 2009. Therefore, the escapement time series for each CU generally ran from 1950 to 2009.

For *trends in abundance* metrics, each assessable CU (and the two CU aggregates) had at least one assessed stream site in the escapement record. Sites were included in the calculation of total EFS for a CU if they were assessed for >70% of the historical time series (Appendix 1 & Appendix 2). An annual EFS record was only included if the field assessment period for that year coincided with peak escapement on the historical record, and if there was a minimum of one site visit. The resolution of the escapement record for a number of CUs changes through time, with many sites that appear in the later time series (increased resolution) rolled up into one site in the early time series (lower resolution). Examining how the resolution of sites changed over time was a critical step in deciding whether a site should be included in the escapement time series for a CU. For example, McNomee Creek (Shuswap-ES) was historically rolled into the Seymour River site records, and was only recently recorded independently from Seymour as its own site. Therefore, in the escapement record there was no data recorded separately for the McNomee site until 1992. If this site was excluded in error, due to a lack of expert knowledge on how the escapement resolution changed over time, this would have introduced a negative bias in the recent time series.

For CUs with either no abundance estimates for any included site in a given year (for CUs with multiple sites), or no abundance estimates for the single included site in a given year (for CUs with only one site), missing data points were gap filled using cycle averages (Appendices 2 & 5). Gap filling is particularly important for dominant cycle years, which, if missing, could significantly reduce the generational mean (i.e., smoothed four year running average) for segments of the time series that include that missing estimate. Using the cycle average method, abundance estimates for any missing year (e.g. 1942) were interpolated by inserting the mean of the same cycle year (4-year cycle) from the immediately previous (e.g. 1938) and subsequent generation (e.g. 1946). If the cycle year of either of the closest two generations was missing (i.e., 4 years previous or subsequent to the missing point), the corresponding cycle year no more than two generations away (e.g. 1934 & 1950) was used to calculate the mean. Interpolation was conducted prior to $\log_e$ transformation and smoothing (with the generational mean) (Appendix 2 & Appendix 5).

For missing data points in CUs with multiple sites, those sites that were spatially proximate and correlated in terms of abundance were grouped together. Gaps within these site groupings were then filled using a mean proportion approach (Appendices 2 & 5). This approach fills gaps based on the proportion each site contributes to the total group abundance, when averaged across years for which data are available for all populations. In addition, for highly cyclic stocks, the gap filling approach separated dominant and subdominant years (and in some cases also weak cycle years), since site proportions varied by cycle year (e.g. for Shuswap Complex-L the dominant and subdominant cycles were gap filled and for Takla-Trembleur-EStu all cycles were separated and then gap filled) (Appendices 2 & 5).

Simulation modelling to compare gap filling methods is currently being investigated at the time of this publication (Carrie Holt, DFO Science).

Recruitment Data

Recruitment data are organized by stock and age by the Pacific Salmon Commission (PSC). Recruitment data are the sum of escapement (see previous section) and catch data, and, in recent years, estimates of Sockeye en-route mortality during upstream migration to natal spawning grounds. Escapement estimation is summarized in the previous section. For catch data, a variety of catch assessment programs are conducted in both marine and freshwaters where fisheries that intercept Fraser Sockeye occur. These fishery monitoring programs are described broadly in integrated fishing plans: *http://www.dfo-mpo.gc.ca/fm-gp/peches-fisheries/ifmp-gmp/index-eng.htm*. Details on study designs and catch statistics can be found for recreational and commercial assessments on the following site: *http://www.pac.dfo-mpo.gc.ca/stats/rec/index-eng.htm*. First Nations catch study designs and statistics are reported on the following site: http://www.pac.dfo-mpo.gc.ca/fraserriver/firstnations.htm. Sockeye catch is partitioned into stock groups by using scale pattern analysis (Gable & Cox-Rogers 1993), and in recent years, supplementary DNA analysis.

For scale pattern analysis, baseline standards are obtained from annual spawning ground sampling (conducted by DFO as part of their annual escapement enumeration programs). Samples are taken from fisheries (by gear type and opening) in Alaska, British Columbia and Washington from late-June through to early October by the PSC. Additionally, Sockeye caught in test fisheries are also sampled in both Panel and Non-Panel waters throughout the fishing season (Gable & Cox-Rogers 1993). In 2000, the Pacific Salmon Commission (PSC) began using DNA analysis, along with scale analysis, to improve the identification of Fraser River sockeye stocks in mixed stock fisheries. Modifications and changes to the stock identification methodology through time, along with the issues associated with each stock identification approach (scale and DNA analyses), add additional uncertainty to the recruitment time series. The PSC presents catch partitioned by stock in their final annual reports (http://www.psc.org/publications_annual_fraserreport.htm) and rolls this catch information up with escapements and also differences between estimates (DBE, described in proceeding paragraph) into their Production (stock-recruitment) database.

Differences between estimates (DBEs) are the estimates of Sockeye abundance as they migrate past Mission (at the Mission Hyroacoustic assessment site) minus the sum of escapement to the spawning grounds and catch (upstream of Mission). Prior to 1992, DBEs were added only when unusual environmental conditions were observed during a stock's upstream migration or spawning periods. Subsequently (1992 to present), differences between estimates were added to a stock's total return using the following decision rules: DBE's were positive (Mission abundance estimates were greater than a stock's catch plus escapement); there was evidence of adverse upstream migration conditions; escapement and/or catch assessments were likely underestimated (biased low), and Mission estimates are unlikely to be significantly biased. Re-evaluation of decision rules for adding run size adjustments (RSAs, given they are not exclusively estimated by subtracting catch plus escapement from Mission hydroacoustic estimates), to total returns (catch plus escapement), is currently part of an on-going Pacific Salmon Treaty Technical Committee process.

For most CUs that have stock-recruitment data, the time series used for status evaluation includes the brood years 1950-2004. Although for most CUs the time series begins prior to 1950, to be consistent with the *trends in abundance* metric only data starting in 1950 were used. Exceptions to this time period include CUs where the stock-recruitment time series was influenced strongly by the introduction of spawning channels, dam blockage, and differences in population dynamics due to hatchery enhancement or poor data. To ensure that the entire stock-recruitment time series is comparable, only brood years that occur after the construction of spawning channels are included in the time series for the Anderson-Seton-ES (brood years 1968-2004), Nadina-Francois-ES (brood years 1973-2004), and Harrison (U/S)-L (brood years

1966-2004) CUs. For the North Barriere-ES (*de novo*) 'CU', only years after the removal of a dam blocking Sockeye access to Fennell Creek were included (brood years 1967-2004).

CUs that have been influenced by hatchery enhancement were also truncated to eliminate enhancement years. In the Shuswap-ES CU, a key site (Scotch Creek) was strongly influenced by hatchery production prior to 1980, therefore, only stock-recruitment data from the 1980-2004 brood years were used. The Seton-L (*de novo*) 'CU' similarly had early hatchery influences and considerable gaps in the early time series, therefore, only stock recruitment data from the 1965-2004 brood years were included. The Cultus-L CU was significantly enhanced in recent years, therefore, the 2001 to 2004 brood years were not included. Although the hatchery program for Cultus-L Sockeye started in 2000, the number of fry produced in this initial year was negligible, therefore stock-recruitment data from the 2000 brood year was included in the time series. Pitt-ES Sockeye stock-recruitment data include adults that were removed for hatchery enhancement, since these fish contribute to subsequent recruitment in this system. Chilko-ES and Chilko-S stock-recruitment data are combined into a single aggregate.

Escapement and Recruitment Data Quality

Overall data quality for each Fraser Sockeye CU can vary amongst sites and years. For the purpose of this paper, both escapement and recruitment data are assigned one of the following five codes to represent data quality:

1) Poor: an estimate with poor accuracy due to poor counting conditions, few surveys (one or two in a given year), incomplete time series, etc.;
2) Fair: an estimate using two or more visual inspections that occur during peak spawning where fish visibility is reasonable; methodology and data quality varies across the time series in terms of good to poor quality;
3) Good: four or more visual inspections with good visibility;
4) Very Good: an estimate of high reliability using mark recapture methods, DIDSON methods, or near-complete fence counts that have relatively high accuracy and precision. Visual surveys that have been calibrated with local fence programs;
5) Excellent: an unbreached fence estimate with extremely high accuracy given an almost complete census of counts.

In general, most Fraser Sockeye escapement data are collected using methods rated three or above in the above (1 to 5) scoring scheme. A more detailed assessment could be conducted in the future to summarize data quality by year and site and across the different data types used (escapement versus recruitment), however, this work was not within the scope of the current paper.

Carrying Capacity Data

Sockeye production can be limited in the freshwater environment by the amount of available spawning ground habitat for egg incubation and/or by the juvenile lake-rearing habitat (food availability, competitor population size, etc.). For Fraser Sockeye, reliable (peer reviewed) estimates of spawning ground capacity currently do not exist. However, juvenile Sockeye lake-rearing capacity estimates are available and are updated in this paper using recent data on photosynthetic rates (PR) and juvenile Sockeye competitor data. These updated capacity data were then used to set Bayesian prior information on lake-rearing habitat capacity (Ricker stock-recruitment model '*b*' parameter priors) for the estimation of *abundance* metric benchmarks, where available and appropriate. Although Bodtker et al. (2007) developed a Bayesian PR

(Photosynthetic Rate) method that explicitly takes into consideration the uncertainty associated with lake productivity-derived estimates of spawner abundance at maximum smolt production (S_{max}), this approach has not been updated in the current paper. Instead, lognormally distributed priors were estimated for the Ricker carrying capacity ('*b*') parameter, using the average lake-specific S_{max} derived from PR models and a standard deviation (sigma) that exceeds the CU's escapement range. River-type CUs do not rear as juveniles in lakes and, therefore, informative juvenile lake rearing capacity priors were not appropriate for these CUs. Some lake-type CUs either did not have PR specific S_{max} estimates, or the S_{max} estimates varied significantly from those estimated from stock-recruitment modelling alone. In these cases where lake-rearing habitat capacity PR estimates are not appropriate for a particular Fraser Sockeye CU, uninformative priors were used (uniformly distributed from 0 to, generally, 1 million).

The current paper updates estimates of juvenile Sockeye lake-rearing capacity, using data on photosynthetic rate (PR) and juvenile Sockeye competitors by lake (Appendix 4, Tables A & B). Photosynthetic rate is positively correlated with fish yield in freshwater lakes (Fee *et al.* 1985; Downing *et al.* 1990) and, in fact, is more closely correlated with fish yield than any other variable (e.g. chlorophyll and total phosphorus) (Downing *et al.* 1990). A Sockeye-specific PR model that predicts the abundance and biomass of Sockeye smolts produced at lake rearing capacity, and the number of spawners required to produce those smolts was developed (Hume *et al.* 1996) by combining PR analysis and the euphotic zone model of Koenings and Burkett (1987). This PR model was recently further revised to explicitly use PR (Shortreed *et al.* 2000) and adjust for the presence of competitors and age-2 smolts (Cox-Rogers *et al.* 2010).

The presence of competitors was expanded in the current paper to consider all common competitors of juvenile Sockeye that are similarly planktivorous (feed on zooplankton). In the Fraser study lakes, the most common competitors are often kokanee (*Oncorhynchus nerka*), and may also include the redside shiner (*Richardsonius balteatus*), threespine stickleback (*Gasterosteus aculeatus*), longfin smelt (*Spirinchus thaleichthys*), Chinook salmon (*O. tshawytscha*), and various whitefish species (*Coregonus* spp.). Based on reports in the literature (Roberge *et al.* 2001; McPhail 2007), and limited stomach analysis (data on file), this analyses assumed that the diet of competitors is the same as that of age-0 Sockeye, and the competitor biomass uses the same proportion of available food as an equivalent amount of Sockeye biomass (Appendix 4, Tables A & B). This is a conservative approach, since our sampling data indicate that these species occupy the lake's limnetic zone and are planktivorous, though we have little data on competitor population variability or diet. Many competitor species may have a wider dietary range than Sockeye, therefore we may be overestimating their competitive overlap for zooplankton prey.

Although data on the abundance, biomass, diet, and temporal variability of juvenile Sockeye competitors are limited, we have made preliminary estimates of competitor biomass based on pelagic surveys. Abundance estimates were derived from hydroacoustic surveys and community composition, and fish size data were obtained from midwater trawling (MacLellan and Hume 2010). In some instances, we were able to distinguish between age-0 Sockeye and kokanee using either genetic or otolith sampling, but these data were not always available. The presence and abundance of age-1 kokanee was inferred from the trawl catch and from the proportion of age-2 smolts in the adult return data (Appendix 3, Table A & B). A considerable amount of work is required to improve these estimates, as sampling was often limited (e.g. 'n/a' in Appendix 3, Table B), and little is known about the seasonal abundance, distribution, or niche overlap of Sockeye competitors in most of these lakes.

In many nursery lakes, a proportion of Sockeye fry from each brood year resides in the lake for more than one year, outmigrating as age-2 smolts. During their lake rearing period, these older fish compete directly with age-0 Sockeye, but they also contribute to smolt production and cannot be treated as simple competitors. While the presence of older smolts will not affect the predicted maximum smolt biomass a lake may produce, they can have a substantial effect on the number of smolts that comprise this biomass. We accounted for the presence of older smolts in our models by using the estimated weighted mean smolt size, based on the proportion of each age class in the smolt run of each brood year (Cox-Rogers *et al.* 2010).

For lakes in the Fraser watershed, the limnological data used in applying the PR model was collected for one to ten years on a monthly basis over most of the growing season (May to October). An exception was Pitt Lake, which was sampled only 3 times over two years (Shortreed *et al.* 2001). A detailed description of the methods used is available in Shortreed et al. (1998). Details of the PR model and the adjustments described are presented in Cox-Rogers et al. (2010).

CLASS OF INDICATORS, METRICS AND BENCHMARKS

Spawner Abundance

For CUs with stock-recruitment data available, Holt et al. (2009) and Holt (2009) recommend using the Ricker stock-recruitment model with a Bayesian approach (with prior information on the carrying capacity parameter where available) to estimate *abundance* benchmarks for Pacific Salmon CUs. For *abundance* metric lower and upper benchmarks, Holt (2009) recommended using, respectively, S_{gen} (the spawner abundance that would result in recovery to S_{MSY} in one generation) and 80% S_{MSY}. Simulation modelling results indicated that using S_{gen} as a lower benchmark was associated with a relatively low probability (<25%) of extirpation over 100 years for populations under equilibrium abundances greater than 15,000 spawners, and a relatively high probability (>75%) of recovery to S_{MSY} in three generations when fishery uncertainties were accounted for (Holt 2009). Details of the Ricker model used to estimate benchmarks for Fraser Sockeye are documented in Holt et al. (2009) and Holt (2009).

In addition to the standard Ricker stock-recruitment model that assumes stationary productivity (constant Ricker 'a' parameter value over the entire time series), recommended by Holt et al. (2009), Ricker model forms that consider time varying productivity were also used to estimate *abundance* benchmarks for Fraser Sockeye CUs. Time varying productivity was an important consideration in benchmark estimation for Fraser Sockeye, since most of these CUs have exhibited systematic declines in productivity over recent decades. Results from simulation modelling that evaluated the probability of extirpation (under a constant escapement policy equal to the S_{gen} lower benchmark) across different assumptions about a population's productivity, reported that extirpation risk was greatest under the assumptions of linear decreases in productivity over time, or constant low productivity, relative to other productivity scenarios (linear increase or stable medium or high productivities) (Holt 2009; Holt and Bradford 2011). In the current paper, several approaches to estimate changes in a CU's intrinsic productivity were used to estimate *abundance* benchmarks by altering the standard Ricker model form. The resulting estimates of productivity better reflect recent periods compared to estimates derived from the standard Ricker model that assumes stationary productivity.

The first approach for estimating time varying productivity was to truncate the stock-recruitment time series to more recent (typically lower productivity) periods when fitting the standard Ricker models. For most CUs with a complete time series (1950-2004 brood years), in addition to estimating benchmarks using the full time series with the standard Ricker model, benchmarks were estimated using two truncated data sets including the brood years 1970-2004 and 1990-2004. For CUs with shorter time series', in addition to using their full time series, data sets were truncated to include only the 1990-2004 brood years in benchmark estimation.

The second approach used to incorporate intrinsic productivity changes in benchmark estimation is the smoothed-Ricker approach. This approach estimates the Ricker '*a*' parameter from a running-average of the last (most recent) four years in the stock-recruitment time series ($a_t = \log_e(R_t/S_t) + bS_t$).

The final (third) approach to estimate time varying productivity is the recursive-Bayesian Ricker model. Similar to the Kalman filter (KF) approach, sometimes used for salmon stock assessment (Dorner et al. 2008), the recursive-Bayesian Ricker model estimates trends in productivity that are due to underlying processes. Estimates of productivity are independent of the interannual variability that can be attributed, at least in part, to observation errors. Although the KF is numerically less demanding than recursive-Bayesian estimation, prior information on model parameters can be easily included into the recursive-Bayesian approach. Including prior information on population capacity (Ricker '*b*' parameter) is especially critical when the time-series of spawner and recruitment data are short or uninformative (most cases). Priors can reduce the biases in parameter estimates that are due to observation errors in spawner abundances ("errors-in-variables") and recruitment anomalies at low spawner abundances ("time-series bias") (Walters and Martell 2004). Here, we build on previous use of recursive-Bayesian Ricker models for estimating time-varying productivity for Sockeye salmon in the Fraser River (Grant et al. 2010), by including informative priors on capacity. In preliminary testing, we found that when priors are uninformative, parameter estimates from the recursive-Bayesian Ricker model converge with the smoothed estimates from a Kalman filter.

Priors on the carrying capacity parameters (see Methods: Carrying Capacity Data; and Results: Carrying Capacity) were incorporated into the different Bayesian Ricker model forms used to estimate *abundance* benchmarks for Fraser Sockeye. Bayesian posterior parameter distributions for the biological models were estimated using WinBUGS (Bayesian software Using Gibbs Sampling) (WinBUGS is available at, http://www.mrc-bsu.cam.ac.uk/bugs/welcome.shtml). Bayesian diagnostics were examined for all models and CUs. We used Gelman & Rubin diagnostics and the Geweke Statistic, G (if G>2 or < -2 then estimates derived from the first 10% of the chain differed from the last 50% and convergence has not occurred), to determine if MCMC chain convergence had occurred. If convergence was not achieved, the number of MCMC trials and/or the burn-in length was increased. Chains were examined for autocorrelation, and were further thinned if this occurred.

The Larkin model, which accounts for the effects of biological interactions among cycle lines, due to, for example, competition for food or predation (Walters and Staley 1987; Cass and Grout 2006; Martell *et al.* 2008), was not included in this current evaluation of uncertainty in *abundance* metric benchmarks. Our current method for estimating benchmarks using the Larkin model assumes that there are equal abundances across each four-year generation of cycle-lines, when incorporating the effect of interactions between cycle lines into the optimization. Though this assumption is necessary for benchmark estimation, it violates a fundamental attribute of the Larkin model, and does not capture the temporal variability in abundances observed for most Fraser Sockeye CUs. Specifically, this assumption is not consistent with the historical persistence of 4-year cycles in CUs with strong delayed-density dependent effects. The assumption of constant abundances across cycle lines for Larkin benchmarks also results in very strong density dependence, low recruitment, and often extremely small S_{MSY} and S_{gen}

values. It may be possible to impose a more realistic (though arguably arbitrary) sequence of spawner abundances into the optimization, or to estimate cycle-line specific benchmarks. Martell et al. (2008) provides an optimization method for assessing options to deliberately maintain cyclic dominance by solving for cycle-line specific optimal escapements that maximize catch where delayed-density dependent effects are significant. Their approach may be useful for assessing Larkin model WSP benchmarks. While cycle-line specific target escapements or harvest rates may be appropriate for management in order to optimize harvest across cycle lines, uncertainty in the biological mechanisms underlying cyclic patterns (and corresponding uncertainty in the probability of extirpation and recovery associated with various cyclic patterns on benchmarks) makes it difficult to justify benchmarks on biological status by cycle line. Instead, a longer-term perspective (e.g. aggregating over cycle lines) may be more appropriate for identifying biological benchmarks and assessing status.

Holt (2009) and Holt and Bradford (2011) found that when the WSP recommended lower benchmark (S_{gen}) was estimated using the Ricker model, the probability of extirpation, and the recovery to S_{MSY} from S_{gen}, were relatively robust to uncertainty in the underlying stock-recruitment dynamics. The performance of lower benchmarks derived from the Ricker model did not vary considerably when the underlying "true" stock-recruitment model was a Larkin model with highly cyclic dynamics, compared to when it was a Ricker model. In contrast, Holt (2009) and Holt and Bradford (2011) found that uncertainty in productivity had a relatively large effect on the probability of extirpation and recovery to S_{MSY} from the lower (S_{gen}) benchmark. Therefore, given the current challenges with estimating Larkin model benchmarks, and given that lower benchmarks estimated using the Ricker model form are relatively robust to the probability of extirpation and recovery to S_{MSY}, only Ricker model derived benchmarks are included in the current paper.

There was one CU (Chilliwack-ES) that did not have stock-recruitment data, but did have alternative data on spawners at maximum juvenile production (S_{max}) (i.e. lake-rearing capacity). For CUs with carrying capacity data available from freshwater production studies, Holt et al. (2009) recommended lower and upper benchmarks of 20% and 40% of S_{max}. For this single CU, deterministic upper and lower benchmarks were, therefore, estimated using this approach.

Holt et al. (2009) recommended evaluating CU status using the *abundance* metric by comparing the geometric mean of the current generation (brood years 2006-2009) of total spawners to the *abundance* benchmarks. A number of Fraser Sockeye populations exhibit highly cyclic annual abundances (four year cycles), often having one persistent dominant (large) abundance cycles, followed by three weaker (lower) abundance cycles. Using a geometric mean on such cyclic, lognormally distributed abundance data, is theoretically most appropriate in representing the average value of these data, because unlike the arithmetic mean, the geometric metric is not inflated by the less frequent, higher abundance years. Technically, this results in a more accurate measure of the central tendency of the abundance data. However, due to the two-tiered enumeration program in place for Fraser Sockeye (see previous Escapement Data section), higher abundance cycles (dominant cycle lines) are generally enumerated with higher precision methods than lower abundance cycles (off-cycle lines). These more accurate and precise estimates are down weighted in the geometric mean approach, giving more relative weight to the less accurate and precise (often biased low), lower abundance estimates. Therefore, the geometric mean may provide a less accurate measure of the centre of data than the arithmetic mean. In this paper, we used both the geometric and the arithmetic mean of the recent generation abundance to evaluate status, to reflect this uncertainty in the most appropriate choice of mean calculations. Holt et al. (2009) also recommended comparing the current year's abundance to the benchmarks, to provide another evaluation for *abundance* CU status. This was not used in the current paper, as status would be highly confounded by cyclic dominance, with evaluations performed on dominant years generally indicating a better status than those performed during weak cycle years.

Trends in Abundance

There are a number of possible metrics within the *trends in abundance* class of indicator, including metrics that compare current abundances (last generation) to a range of historical baselines (e.g. historical average, a historical maximum, first generation in the time series), and metrics that measure trends (e.g. linear change in abundance over the last three generations). A recent study evaluated the effectiveness of different metrics in correctly categorizing the *trends in abundance* status using Fraser Sockeye abundance data for 18 CUs, using a Receiver Operating Characteristic (ROC) approach and retrospective analysis (Porszt 2009). In the current paper, the Porszt (2009) study has been updated to include two additional metrics: the ratio of the geometric mean spawner abundance of the current generation to the historical mean, and to the mean of the first three generations. These analyses concluded that metrics that generally ranked high in identifying 'true' status were those that compared the last generation abundance to historical baselines (e.g. time series average). Metrics that identified status by comparing the last generation abundance to the historical maximum consistently ranked low, and metrics that evaluated linear changes in abundance over the last three generations performed intermediate amongst all metrics evaluated.

Three metrics were chosen to assess *trends in abundance* for each CU, based on the toolkit of metrics presented by Holt et al. (2009), results from Porszt (2009), and the recent evaluation of several additional metrics (described above). The first *trends in abundance* metric examines changes in abundance over the long-term, using the ratio of the current generation geometric mean to the long-term geometric mean. The final two metrics evaluate trends over the short-term, measuring the linear change in abundance over the most recent three generations using both a deterministic and probabilistic approach, as described in Holt et al. (2009). The long-term trends in abundance metric (ratio of the current generational geometric mean to the long-term geometric mean) was estimated using the smoothed, $\log_e$ transformed EFS time series for each CU. The time series' were first log transformed then smoothed by taking a 4-year running average starting on the second year of data in a time series, so that the average for each year includes the one year before and two years after that year. The purpose of this transformation is to minimize the influence of cyclic abundances and observation and assessment errors on trend status evaluations. Lower and upper benchmarks for this metric described by Holt et al. (2009) were, respectively, ratios of 0.25 and 0.5. However, in a previous publication, Pestal and Cass (2009) considered ratios of less than 0.5 as low in terms of status, and those above 0.5 as ranging from below average to above average in status (Petal and Cass 2009). Therefore, in the current paper we used 0.5 as the lower benchmark (ratios below this value are considered low to very low: Red status), and 0.75 as the upper benchmark (ratios above this value are considered near or above average: Green status) to assess status for the ratio metric. Recent to long-term average ratios between 0.5 and 0.75 are considered average (Amber status).

For *recent trends in abundance*, the linear change in abundance over the last three generations was estimated both deterministically and probabilistically (two metrics). The Committee on the Status of Endangered Wildlife in Canada (COSEWIC) and the International Union for the Conservation of Nature (IUCN), both use the change in abundance over the last three generations (or 10 years, whichever is longer) as their *trends in abundance* metric (COSEWIC 2003; Rand 2008) to assess wildlife status. To calculate the last three generation trend, the abundance time series' were first converted to $\log_e$ space, then smoothed using the four year running average, to remove the annual "noise" that obscures underlying trends in population abundance (COSEWIC 2003). Regression analyses were conducted on the last three generations (1998-2009) of the transformed abundance data to calculate the linear change in abundance. The slope, calculated in $\log_e$ space, was then compared to the upper and lower benchmarks presented in the proceeding paragraph, to assess status by CU on this metric.

The upper and lower benchmarks for the recent trends in abundance metric, as used by Holt et al. (2009), are respectively, a 15% decline and a 25% decline in abundance. The lower benchmark represents a smaller reduction in abundance (more biologically conservative) than the COSEWIC/IUCN guideline (30% reduction) that classifies a species as "threatened". To apply these changes in abundance as benchmarks, the linear rates of change that are associated with a reduction of 25% (lower benchmark) and 15% (upper benchmark) over three generations (i.e. the slope of a line of best fit that relates to a decline of this size) were used. These declines (15% and 25%) convert to slopes (in $\log_e$ space) of, respectively, -0.015 (upper benchmark) and -0.026 (lower benchmark). These $\log_e$ slopes deviate slightly from those used by Holt et al. (2009), due to a modification in the slope calculation. Specifically, the slope calculation used by Holt et al. (2009) examined the abundance changes over 12 years, while the current paper used 11 years, related to the change (in years) from year 1 (e.g. 1998) to year 12 (e.g. 2009). The second recent trends in abundance metric explores uncertainty in the linear change in abundance over the last three generations ($\log_e$ slope as described above) by calculating the probability that the slope is below the lower benchmark of 25% (slope: -0.026). This metric is not independent of the previously described recent trends in abundance metric, but rather is complementary, providing additional support to the deterministic recent trends in abundance status.

Productivity

Productivity indices combine the freshwater and marine mortality data presented in Grant et al. (2010), and have been updated to reflect the current CU level of organization. The three indices include $\log_e$(R/EFS), Ricker model residuals, calculated as deviations between the model's annual predictions and observations (Ricker 1975), and smoothed time varying Ricker model '*a*' (intrinsic productivity) parameter values, estimated annually using a recursive-Bayesian procedure (KF Ricker '*a*' parameter) (Peterman *et al.* 2000; Peterman *et al.* 2003; Dorner *et al.* 2008)(Appendix 3, Figures 1, c & d). The $\log_e$(R/EFS) productivity index describes total CU productivity. The remaining two indices remove the density dependent effects of spawner abundance in the total $\log_e$(R/EFS) variability. The KF Ricker '*a*' parameter values further remove short-term variability in $\log_e$(R/EFS) productivity.

RESULTS: WILD SALMON POLICY STATUS EVALUATION

CONSERVATION UNIT BIOLOGICAL STATUS

A*bundance* and *trends in abundance* metrics were used to evaluate statuses for 22 current and two *de novo* Fraser Sockeye CUs. Detailed descriptions of analyses and results for each CU, including data used, historical background, and status assessments, are reported in the proceeding sections.

Carrying Capacity

To update estimates of spawner capacity at maximum juvenile production (S_{max}) in the current paper, lakes were placed into one of three groups depending on the extent to which their capacity is influenced by competitor populations (Appendix 3, Tables A & B). In the first group of lakes (e.g. Bowron, Chilko, Francois, Kamloops, and Lillooet Lakes), characterized by the absence of non-Sockeye in catch samples, productive capacity was not measurably reduced by competitors. In the second group of lakes (e.g. Cultus, Adams, Fraser, Mabel, Trembleur, Shuswap, and Quesnel Lakes), characterized by high variance in non-Sockeye catch samples, moderate reductions (1-10%; mean ~6%) in productive capacity were estimated due to competitor foraging. A third group of lakes (Anderson, Chilliwack, Harrison, Seton, Pitt, Stuart, and Takla Lakes), characterized by high variance in non-Sockeye catch, were associated with a large reduction in productive capacity by competitors (15-90%; mean ~37%). Maximum spawner capacity for lakes within each of these three groups was estimated by applying a mean competitor adjustment within groups (Appendix 4, Table C). These updated S_{max} estimates were then used as the mean estimates for lognormally distributed priors for the Ricker carrying capacity ('*b*') parameter (Table 3). Standard deviations (sigma) used for these priors were set to exceed the CU's escapement range (Table 3). See preceding Methods section describing the decision to include either informative lognormally distributed priors (using updated S_{max} estimates for their means) or uninformative uniformly distributed priors for Ricker carrying capacity ('b') parameters, used in *abundance* benchmark estimation for CUs (Table 3).

Given the limitations inherent in the available competitor data, and the assumptions that were necessary in order to develop biomass estimates, we cannot assign a high degree of confidence to the estimates of competitor biomass. Therefore, estimates should be used with caution, and with a full understanding of how they were derived. For lakes in the Fraser watershed that were included in the PR model, but for which we were unable to develop an estimate of competitor biomass, it may be appropriate to assign a value derived for other lakes with similar ecologies and species compositions. For example, smelt are known to be abundant in the pelagic zone of Pitt Lake in a similar fashion to Harrison Lake (Henderson *et al.* 1991). Thus, we thought it reasonable to assign Pitt Lake to same group as Harrison Lake.

Table 3. Lake-rearing capacity (number of spawners that result in maximum juvenile production: S_{max}) estimates used as carrying capacity (Ricker 'b' parameter) Bayesian priors in the abundance benchmark estimation process. The first column presents the CU, the second column presents the range of the stock-recruitment (SR) time series, the third column presents the prior distributions used (uniform or lognormal), the forth column presents either the average for the lognormal prior distributions or the range for uniform distributions and the final (fifth column) presents the sigma used for the lognormal prior distributions.

CONSERVATION UNIT	SR Time Series (Brood Years)	Spawning (S_{max}) Capacity Used in SR Models			S_{max}: Lake Rearing[1]		
		Prior Distribution	Average	Sigma	Average	SD	*N*
Anderson-Seton-ES	1968-2004	Uniform	0-1,000,000	NA	286,000	54,000	4
Bowron-ES	1950-2004	Lake Rearing	40,000	13,000	40,000	NA	1
Chilko-S & Chilko-ES	1950-2004	Lake Rearing	400,000	16,000	483,000	161,000	6
Cultus-L	1950-2000	Lake Rearing	80,000	12,000	85,000	17,000	3
Francois-Fraser-S	1950-2004	Lake Rearing	550,000	13,000	600,000	201,000	2
Harrison (U/S)-L	1966-2004	Uniform	0-1,000,000	NA	811,000	316,000	2
Harrison River (River-Type) (immediate migrants)	1950-2004	Uniform	0-800,000	NA	NA	NA	
Kamloops-ES	1967-2004	Uniform	0-500,000	NA	445,000	NA	1
Lillooet-Harrison-L	1950-2004	Lake Rearing	164,000	13,000	164,000	NA	1
Nadina-Francois-ES	1973-2004	Uniform	0-1,000,000	NA	1,350,000	453,000	2
North Barriere-ES (*de novo*)	1967-2004	Uniform	0-50,000	NA	NA	NA	
Pitt-ES	1950-2004	Uniform	1-1,500,000	NA	115,000	NA	1
Quesnel-S	1950-2004	Lake Rearing	1,000,000	12,000	1,115,000	315,000	10
Seton-L (*De Novo*)	1965-2004	Uniform	0-300,000	NA	188,000	31,000	4
Shuswap-ES	1980-2004	Uniform	0-2,000,000	NA	1,900,000	319,000	6
Shuswap Complex-L	1950-2004	Lake Rearing	1,500,000	15,000	1,900,000	319,000	6
Takla-Trembleur-EStu	1950-2004	Lake Rearing	600,000	15,000	778,000	165,000	3
Takla-Trembleur-Stuart-S	1950-2004	Lake Rearing	1,400,000	16,000	1,900,000	193,000	3

1. Source: J. Hume & L. Pon, Salmon Aquatic Freshwater Ecosystem Program, DFO; Appendix 3.

Abundance metric benchmarks and status

For each CU with stock-recruitment data (19 CUs in total, including the aggregated Chilko-ES/Chilko-S CU), benchmarks for *abundance* metrics were estimated using the Ricker stock-recruitment model and a Bayesian approach, including forms that assume stationary and non-stationary productivity. Specifically, benchmarks were estimated using the standard Ricker model with the full stock-recruitment time series that assumes stationary productivity through time and Ricker model forms that assume non-stationary productivity including the Ricker model with truncated (more recent) stock-recruitment time series', a smoothed-Ricker model (recent four year running averages on the intrinsic productivity Ricker *a* parameter), and a recursive-Bayesian Ricker model (Table 4). For all model forms, benchmarks were presented across six probability levels (10%, 25%, 50%, 75% and 90%) to reflect the stochastic uncertainty in the model fit to stock-recruitment data (Table 4). To assess status for each model and probability level combination by CU, recent average CU abundances, estimated using both arithmetic and geometric means, were compared to each of the paired lower and upper benchmarks (Table 4). One CU (Chilliwack-ES) did not have stock-recruitment data but instead had carrying capacity data which were used to estimate deterministic lower and upper *abundance* metric benchmarks. Therefore, in total, *abundance* metric statuses were assessed for 20 CUs.

Across all Ricker model forms explored, generally the standard Ricker model (using the full stock-recruitment time series) that assumes stationary (constant) productivity produced the smallest (least biologically conservative) lower benchmarks (Table 4). Standard (full time-series) Ricker models assume that productivity is constant over the entire time series, despite the changes in productivity exhibited by most CUs over time (e.g. generally high at the start of the time series and systematic declines in recent decades)(Appendix 3, Figures 1 c & d). In contrast, Ricker model forms that assumed non-stationary productivity (e.g. the Ricker model fit to the most truncated stock-recruitment time series (brood years 1990-2004), the smoothed and recursive-Bayesian Ricker models) generally produced the highest (most biologically conservative) lower benchmarks (Table 4). One major exception to this lower benchmark pattern is the Harrison River (River-Type) CU, which has exhibited the highest productivity of its stock-recruitment time series in recent years (Appendix 3, Harrison River (River-Type), Figures 1 c), and therefore, produced smaller lower benchmarks when models that specifically consider this recent productivity period were used (Table 4). For most CUs, in contrast to the observed pattern in lower benchmarks, the upper benchmarks do not exhibit similar increases when using model forms that emphasize recent productivity, compared to the standard (full time-series) Ricker model (Table 4).

In addition to presenting structural uncertainty in *abundance* benchmarks, through the use of different Ricker model forms to estimate benchmarks, stochastic uncertainty was also explored by presenting benchmarks for each model form across five probability levels (10%, 25%, 50%, 75%, and 90%) (Table 4). Lower benchmark values (less biologically conservative values) occur at lower probability levels and higher benchmark values (more biologically conservative values) occur at higher probability levels. Comparisons of status across model forms and probability levels provide an indication of how sensitive *abundance* metric statuses are to these types of uncertainty for each CU. For example, for eight CUs (Takla-Trembleur-EStu, Bowron-ES, Nadina-Francois-ES, Pitt-ES, Takla-Trembleur-Stuart-S, Quesnel-S, Cultus-L, and Harrison River (River-Type)), *abundance* metric statuses were relatively insensitive to the model or probability level (i.e. statuses were relatively constant across models and probability levels). For most other CUs, however, statuses were sensitive to the model form and the probability level (i.e. status varied across models and probability levels)(Table 4).

Differences between arithmetic and geometric means of the last generation (brood years 2006-2009) for each CU were generally less than 30%. Major exceptions include Shuswap-ES, Chilliwack-ES, Shuswap Complex-L, and Seton-L (*de novo* 'CU'), for which the differences between the arithmetic and geometric mean calculations were greater than 60%. Since these CUs exhibited large variations in abundances over the last generation, the geometric mean down weights the larger abundance years, and, therefore, produces a much lower recent abundance estimate relative to the arithmetic mean. As a result, *abundance* metric statuses for these four CUs are quite different depending on whether the arithmetic or geometric mean is compared to the *abundance* benchmarks. Specifically, for these CUs, use of the geometric mean results in poorer statuses across models and probability levels relative to the arithmetic mean. For most other CUs, the differences in *abundance* metric statuses across models and probability levels are not significantly different when arithmetic versus geometric means are compared.

Trends in Abundance Metrics and Status

Statuses for three *trends in abundance* metrics were evaluated for each CU. One metric considered long-term trends in abundance (ratio of the current generation geometric mean to the long-term geometric mean) and two metrics considered recent trends in abundance (linear change in abundance in the last three generations, and the probability that this change is below the lower benchmark for this metric). The probabilitistic recent trends in abundance metric is not independent of the deterministic recent trends in abundance metric, but instead gives further weight to the status of the deterministic metric. Of the 22 current CUs and two *de novo* 'CUs', the Chilliwack-ES could not be assessed for *trends in abundance* status, given that the time series was too short to conduct these analyses. In addition, two CUs (Chilko-ES and Chilko-S) were aggregated into one for *trends in abundance* analyses, since their time series could not be disaggregated due to escapement enumeration methods. Therefore, there are 22 *trends in abundance* assessments conducted in the current paper (Table 5).

Of these 22 CUs (including the Chilko-ES & Chilko-S aggregate) where *trends in abundance* statuses were assessed, 50% (10 out of 22 CUs) exhibited long-term trends in abundance statuses that were in the WSP Green zone and recent trends in abundance statuses that were in the WSP Red zone (Table 5). An additional two CUs were Amber in status for long-term trends and Red in status for recent trends (Takla-Trembleur-EStu and Nahatlach-ES). These status trends are generally related to the above average abundances exhibited by these CUs in the 1990's, and the subsequent abundance declines in recent years as these CUs returns to average abundances (Table 5).

One exception to these 12 CUs that exhibited Green (or Amber) long-term trends in abundance statuses and Red recent trends in abundance statuses, was the Shuswap-ES CU. In particular, the recent trends in abundance status of Red for Shuswap-ES initially appear to be counter-intuitive, as arithmetic means indicate that this CU has increased, not decreased, in abundance over the past three generations (Table 5). However, since geometric means are used specifically in trends in abundance status evaluations, when geometric means are compared between the generation third to last (12,700) versus the last generation (7,700), this CU has decreased in abundance. The difference between the arithmetic and geometric means for Shuswap-ES can be attributed to a period (1993-2001) of decreased cyclicity that encompasses the generation third from last (1993-2001) during which dominant cycles exhibited lower abundances and weak cycles exhibited higher abundances, relative to typical cycle years. This period of lower cyclicity produces higher geometric means than the typical Shuswap-ES pattern of abundance (one dominant and three weaker cycles), driving this CUs declining trend in the last three generations.

Divergences from this general pattern in the *trends in abundance* status across metrics include the following: six CUs (Kamloops-ES, Pitt-ES, Harrison River (River-Type), Shuswap Complex-L, Harrison (D/S)-L and Lillooet-Harrison-L) were in the WSP Green zone for both recent and long-term trends in abundance metrics; three CUs (Bowron-ES, Taseko-ES, and Cultus-L) were in the Red zone for all *trends in abundance* metrics; one CU (Widgeon (River-Type)) was in the WSP Red zone for the long-term trends in abundance metric and WSP Green zone for the recent trends in abundance metrics (Table 5).

Current (22) and De Novo (2) Conservation Units

Twenty-four CUs have sufficient information to explore the uncertainty in statuses across metrics and benchmarks. Of these twenty-four, Chilko-ES cannot be independently assessed, as its data are rolled up with Chilko-S. There are two *de novo* 'CUs' that have originated from hatchery transplants and, therefore, are not WSP CUs. These *de novo* 'CUs' are included for reference but technically should not be included in a WSP CU list. Details and results are presented in the proceeding section by CU.

Anderson-Seton-ES

Sites: Populations that rear in Anderson Lake include Gates Creek and Gates Channel (Appendix 1).

History: There is evidence (e.g. fry and smolt outmigration assessments and scale freshwater circuli growth patterns) that Gates Creek and Channel fry rear in both Seton and Anderson Lakes (Geen and Andrew 1961; Roos 1991). As a result, this CU is named after these two rearing lakes.

Between 1919 and 1930, over 15 million Sockeye eggs and juveniles were transplanted to Gates Creek, Gates Lake and Anderson Lake from the Birkenhead River and Sweltzer Creek. An additional transfer of fry occurred in 1950 from the Adams River to Anderson Lake (Aro 1979). The resulting current Anderson-Seton-ES population is considered genetically distinct (Withler *et al.* 2000), and its low genetic diversity and unusual allele frequencies reflect founder effects and/or genetic drift at small population sizes (Withler *et al.* 2000).

The natural spawning area of Gates Creek historically supported an estimated 150,000 Sockeye. However, forest harvesting and the encroachment of human activities are believed to have deteriorated habitat quality, and restricted Sockeye production to the point where only 10,000 Sockeye could be accommodated by the late 1960's (Doug Lofthouse, Oceans, Habitat & Enhancement Branch, DFO, pers. comm.). As a result, between 1967 and 1968, the Gates Creek Sockeye spawning channel was constructed at the west end of Anderson Lake, to compensate for lost production from Gates Creek and Anderson and Seton Lakes. The channel is estimated to account for a high proportion of the CU's production. Gates Channel has an available spawning area of 11,300 m^2, and was designed to accommodate 18,000 Sockeye (D. Lofthouse, DFO, pers. comm.).

A hydro facility on this system has been operational since 1956 (Roos 1991). This facility is comprised of the Seton Dam, located below the outlet of Seton Lake, and the Cayoosh Dam on Cayoosh Creek. Water is diverted by canal from Seton Lake to a powerhouse on the Fraser River, where it is released through a tailrace located 500 m downstream of the outlet of Seton River. Since the Seton Dam presents a barrier to Sockeye migration, a fishway was constructed in concert with dam construction (Roos 1991). It has been reported that both the tailrace and fishway may slow or impede Sockeye migration and cause physiological stress to the fish(Roscoe and Hinch 2008). Due to the downstream tailrace location, migrating adult Sockeye have been shown to stop at the outlet of the tailrace, where that are either attracted to the home-stream water or they use it as a cold-water refuge. Fish may either be directly injured in the tailrace (Fretwell 1980) or indirectly suffer pre-spawn mortality due to the delay in migration from stalling at the tailrace. Success of fish departing the tailrace, entering the Seton River, and reaching the dam depends on Seton water quality, whereby higher Cayoosh Creek dilution results in higher migration failure (10-30% migration failure during IPFSC studies). Once fish enter the Seton River they must travel five kilometres upriver, ascend the Seton Dam fishway, and then migrate through Seton Lake and Anderson Lake (~50km) to the spawning grounds.

One study indicated that locating the fishway entrance presents a challenge to migrating Sockeye (during experimental downstream transplants 25% of these Sockeye could not re-locate the fishway entrance) (Roscoe and Hinch 2008). Further impacts of the hydro facility include mortality (~10%) of downstream migrating smolts as they move through the dam turbines. This issue has yet to be resolved (Roos 1991).

Escapement Time Series: Two sites are included in the escapement time series: Gates Creek and Gates Channel (Appendix 1). Gates Creek was consistently assessed starting in 1954, using peak live cumulative dead visual survey methods up to 1979, with the exception of 1964 when a mark recapture assessment was conducted. Starting in 1980, the creek was assessed using counts of Sockeye diverted into the creek at the diversion weir. Given the public location of the diversion weir, vandalism has compromised the escapement time series of Gates Creek and, therefore, post-1980 these are likely minimum escapement estimates. Gates Channel (operations commenced in 1968) was assessed throughout the time period using a census of carcasses recovered in the channel. The Gates Creek and Channel sites were combined to evaluate status. They cannot be evaluated independently, since numbers of Sockeye distributed between the channel and creek are a consequence of loading regimes at the outlet of this system (Roberta Cook, Ocean Habitat Enhancement Branch, DFO). No gap filling was required for this time series (Appendix 2).

Productivity: Similar to other Early Summer Run and Early Stuart CUs, Anderson-Seton-ES has exhibited systematic declines in productivity (Kalman filter Ricker *a* parameter values) since the late-1960 brood years (Appendix 3, Anderson-Seton-ES, Figure 1 c). Productivity (R/S) has been particularly low during the most recent brood years (1998 to 2005), with four of these years having productivities that are below replacement (Appendix 3, Anderson-Seton-ES, Figure 1 d). Similar to other CUs with freshwater survival data, Anderson-Seton-ES early freshwater survival (fry/EFS) decreased consistently from the start of the time series in 1968 to the mid-1990 brood years, and has subsequently increased (Appendix 3, Anderson-Seton-ES, Figure 1 e). Post-fry survival (recruits/fry), that includes a period of freshwater survival and marine survival, decreased post-channel construction (1969) (Appendix 3, Anderson-Seton-ES, Figure 1 f).

Abundance: The stock-recruitment time series for Anderson-Seton-ES was truncated to include only years after the construction of the spawning channel (brood years 1968-2004). This ensures consistency in the production time series and spawning area. For Ricker model benchmark estimates (recommended model by Holt et al. 2009), a uniformly distributed prior (range: 0 to 1,000,000) was used to estimate the Ricker '*b*' parameter (Table 4; Appendix 3, Anderson-Seton-ES, Figure 2 d). Using the standard Ricker model and the full time series (brood years 1968-2004), lower benchmarks ranged from 1,000 to 9,000 at the 10% to 90% probability level (p-level), and upper benchmarks ranged from 12,000 to 48,000 (Table 4; Appendix 3, Anderson-Seton-ES, Figure 2 e). Statuses were assessed by comparing both the arithmetic (4,100) and the geometric (3,300) mean abundance of the last generation to the benchmarks. The resulting statuses decreased at higher probability levels. Specifically, status changed from Amber to Red above the 75% and 50% p-levels when evaluated using, respectively, the arithmetic and geometric mean abundances (Table 4). Statuses were similar using these two calculations of recent abundance, as geometric and arithmetic recent generation means were similar (only a 20% difference between arithmetic and geometric means).

Given that Anderson-Seton-ES has exhibited systematic declines in productivity, model forms that specifically consider recent lower productivity in benchmark estimation produced higher (more biologically conservative) lower benchmarks. The recursive-Bayesian Ricker benchmarks are not recommended for this CU, as they produce S_{max} estimates that are unrealistically high relative to other models. The most truncated time series (brood years 1990-2004) Ricker model produced the highest lower benchmarks, followed by the smoothed-Ricker version. Upper benchmarks were higher for the truncated time series Ricker model and lower for the smoothed-Ricker model, compared to the full time series standard Ricker model. Statuses for these model forms that consider recent productivity decreased identically at higher probability levels; status changed from Amber to Red above the 25% p-levels (Table 4).

Trends in Abundance: The early time series prior to channel construction is characterized by lower spawner abundances (arithmetic average EFS from 1954 to 1974: 1,300) (Appendix 3, Anderson-Seton-ES, Figures 1 a & b). Escapements (EFS) increased starting in the 1970's (EFS 1970-2009 average: 4,500), coinciding with channel construction. This CU has recently declined from a period of above average EFS, which occurred three generations prior to the end of the time series (6,200), to the current generation average EFS (2,400) (Table 5; Appendix 3, Anderson-Seton-ES, Figure 1 b). This CU exhibits strong cyclic dominance throughout the time series (one dominant cycle average EFS: 8,300; three weak cycles average EFS: 2,100). Generally, spawner success on the time series has been high (70%), with the exception of more recent years. Spawner success dropped between 1995 and 2002, to an average of 56%. Years when spawner success was particularly low include 1992 (channel: 37% & creek: 50%), 1996 (channel & creek: 25%), 2000 (channel: 32% & creek: 47%), 2001 (creek only: 49%), and 2008 (channel and creek: 23%) (Appendix 3, Anderson-Seton-ES, Figure 1 b).

The ratio of the recent generation geometric mean relative to the long-term geometric mean for Anderson-Seton-ES EFS (ratio: 1.98) is more than double the upper benchmark for this metric (ratio: 0.75) (Green status) (Table 5; Appendix 2, Anderson-Seton-ES, Figure 2 c). For comparison, if only data after the installation of Gates channel (1968-2009) are used to estimate the trend in abundance, the ratio of the recent generation abundance to the long-term average would still be greater (Green status) (ratio: 1.37) than the upper benchmark for this metric (ratio: 0.75). In recent years (last three generations), Anderson-Seton-ES EFS has decreased following a period of above average EFS (see previous paragraph). The slope of this recent trend (-38% change in abundance) is steeper than the lower benchmark for this metric (-25% change in abundance)(Red status), and there is an 80% probability that this recent trend falls below the lower benchmark (Table 5; Appendix 3, Anderson-Seton-ES, Figures 2 a & b).

Bowron-ES

Sites: The populations that rear in Bowron Lake (Early Summer timing) include Bowron River, Pomeroy, Huckey, and Sus Creeks, and may also include Antler Creek (see Escapement Time Series section below) (Appendix 1).

History: Hatchery transplants were introduced into the Bowron system from Lakelse Lake (Skeena River hatchery) between 1924 and 1926 (Aro 1979). Since these transplants were not successful, population expansion within this CU after the Hells Gate landslide is likely attributed to remnant Bowron-ES Sockeye (Withler 1982). There was a significant Mountain Pine Beatle outbreak in the 1980's in the Bowron watershed that resulted in significant forest harvesting in this area (K. Peters, DFO, pers. comm.).

Escapement Time Series: Four sites were included in the Bowron-ES escapement time series: Bowron River, Huckey, Pomeroy and Sus Creeks (Appendix 1). For early years in the escapement records, the Bowron River time series includes Pomeroy, Sus and Huckey Creek estimates; whereas in recent years there are a few independent assessments for these smaller creeks (Appendix 2). In years when Huckey, Pomeroy and Sus were assessed independently, their contribution to total production of the CU was 0 (Pomeroy & Sus) to negligible (Huckey); Bowron River dominates total production. Escapement enumeration methods varied from largely fence counts in the earlier time series (1950-1963) to largely visual surveys (helicopter) from 1964 to present, with no gaps in the time series (Appendix 2). In 1995, a fence was installed to re-evaluate the expansion factor used to calibrate the visual surveys in this system. It was found that the expansion factor appropriate for this system (2.9) is much higher than that typically used for Fraser Sockeye (1.8). Therefore, previous surveys (1985-1994) may underestimate true escapement (Schubert 2007).

Antler Creek was excluded from the time series given the limited numbers of years it was assessed (only from 1950-1961) during opportunistic surveys from a fence program, and its small contribution to overall abundance in this CU (~1% of total escapement). Although unconfirmed, Antler Creek Sockeye may not rear in Bowron Lake. Given that fry from these Sockeye spawners would have to travel upstream through fast flowing conditions to reach the lake; these Sockeye may actually be river-type. Recently, Sockeye have been observed in the upper Bowron River, and similarly, these also may be a river-type population (See proceeding Validation Required section on Upper Fraser (River-Type)).

Productivity: Similar to other Early Summer Run and Early Stuart CUs, Bowron-ES has exhibited systematic declines in productivity (Kalman filter Ricker '*a*' parameter values) since the mid-1960 brood years (Appendix 3, Bowron-ES, Figure 1 c). Productivity (R/S) has been particularly low recently (1994 to 2005 brood years), with six of these years close to or below replacement (Appendix 3, Bowron-ES, Figure 1 d). There are no freshwater or marine survival data available for this CU.

Abundance: The full stock-recruitment time series for Bowron-ES includes the brood years 1950-2004. For Ricker model benchmark estimates (recommended model by Holt et al. 2009), a lognormally distributed prior (mean of 40,000 and sigma of 13,000), based on calculations of lake rearing capacity, was used to estimate the Ricker '*b*' parameter (Table 4; Appendix 3, Bowron-ES, Figure 2 d). Using the standard Ricker model and the full time series (brood years 1950-2004), lower benchmarks ranged from 3,000 to 6,000 at the 10% to 90% p-levels, and upper benchmarks ranged from 13,000 to 22,000 (Table 4; Appendix 3, Bowron-ES, Figure 2 e). Statuses were assessed by comparing both the arithmetic (1,600) and the geometric (1,500) mean abundance of the last generation to the benchmarks. The resulting statuses were Red across all p-levels (Table 4). Statuses were similar using these two calculations of recent abundance, as geometric and arithmetic recent generation means were similar (only a 6% difference between arithmetic and geometric means).

Given that Bowron-ES has exhibited systematic declines in productivity, model forms that specifically consider this recent lower productivity in benchmark estimation produced higher (more biologically conservative) lower benchmarks. The most truncated time series (brood years 1990-2004) and the recursive-Bayesian Ricker models produced the highest lower benchmarks, followed by the smoothed-Ricker model. Upper benchmarks were higher when using the truncated time series and the recursive-Bayesian Ricker models, and lower for the smoothed-Ricker model, compared to the full time series standard Ricker model. Statuses for these model forms that consider recent productivity were also identical (Red status) across all p-levels (Table 4).

Trends in Abundance: Bowron-ES exhibited relatively high escapements (EFS) early in the time series (1950-1959 EFS average: 7,400) relative to the time series average (4,300) (Appendix 3, Bowron-ES, Figures 1 a & b). This CU has declined in abundance, from an average of 3,900 EFS, which occurred three generations prior to the end of the time series, to the current generation average of 800 EFS (Table 5; Appendix 3, Bowron-ES, Figure 1 b). From 1959 to 1979, the CU exhibited strong cyclic dominance (one dominant cycle average EFS: 13,600; three weak cycles average EFS: 1,600). Cyclic dominance subsequently disappeared (1983-2009 average EFS: 3,200). Spawner success has remained high throughout the time series (~91%) and has not exhibited any persistent trends (Appendix 3, Bowron-ES, Figure 1 b).

The ratio of the recent generation geometric mean relative to the long-term geometric mean for Bowron-ES (ratio: 0.27) is below the lower benchmark for this metric (ratio: 0.5) (Red status) (Table 5; Appendix 2, Bowron-ES, Figure 2 c). In recent years (last three generations), Bowron-ES EFS has decreased, following a period of average EFS (see previous paragraph). The slope of this recent trend (-90% change in abundance) is steeper than the lower benchmark for this metric (-25% change in abundance) (Red status), and there is a 99% probability that this recent trend is below the lower benchmark (Table 5; Appendix 3, Bowron-ES, Figures 2 a & b). This recent trend is likely more pronounced, given that the early observed abundance time series is quite possibly biased low (see preceding section on Escapement Time Series).

Chilko-ES and Chilko-S (CUs combined for status assessment)

Sites: Populations that rear in Chilko lake include Chilko River, South End of Chilko Lake, North End of Chilko Lake and Chilko River Channel (Appendix 1).

History: Chilko Lake is a large oligotrophic lake far from any significant human development in the Fraser River watershed. The south end of the lake is surrounded by glaciated mountains, and the northern portion extends onto the edge of the interior plateau of BC. Due to its glacial influence, this lake has historically experienced cooler temperatures. Several glacially turbid rivers enter the southern half of the lake, causing water clarity to decrease from north to south during the summer months. The lake's orientation and proximity to the Coast Mountains result in frequent strong southerly winds. As a result, the lake has a cool epilimnion and an unstable thermal regime.

Amongst populations with similar run timing that spawn upstream of Hells Gate, Chilko Sockeye were the least impacted by the 1913 Hells Gate landslide, despite the fact that Chilko Sockeye migration has almost double the grade (twice as steep) of any other Fraser River CUs. The limited impact of the Hells Gate landslide on Chilko Sockeye, relative to other Sockeye CUs, is hypothesized to be linked to their greater energy reserves and their ability to therefore withstand delays in migration (Roos 1991). In recent studies, Chilko Sockeye (relative to other similar timed Fraser Sockeye CUs) have been identified as superoptimal migrants, having greater stride lengths, higher ground speed per tail beat, and lower energy usage than would be predicted (Hinch and Rand 2000). Chilko Sockeye are more torpedo shaped than other Sockeye CUs, which would enhance water flow over the body and decrease drag. As a result, Chilko Sockeye have migration advantages over other similar timed Fraser Sockeye CUs.

Chilko Lake was fertilized in 1988, and again during 1990-1993. Bradford et al. (2000) reported that the size of smolts increased during these periods of fertilization. They also found a positive correlation between the larger smolt body sizes and smolt-to-adult (marine) survival. Fertilization also appeared to have increased abundance on the weaker 1989 cycle, and improved survival during the early 1990's, when productivity for most other CUs decreased (Appendix 3, Chilko-ES & Chilko-S, Figures 1 a-f) (Bradford *et al.* 2000). Limnological surveys, conducted in 2009, in response to recent increases in smolt production, found that the photosynthetic rate had increased to rates similar to those seen during fertilization (D. Selbie, DFO, pers. comm.). In addition to fertilization, a small artificial side channel was operated from 1988 to 2004 on Chilko River, to enhance the productive capacity of Chilko; although spawning habitat did not appear to be limiting to Sockeye at that time. Post-2004 this channel was decommissioned, and, therefore, became inaccessible to Chilko Sockeye.

Escapement Time Series: All sites were included in the escapement time series since they represent one complete time series with the North and South End of Chilko Lake assessed separately in some years and in other years included in the Chilko River site in the escapement database (Appendix 1). The Chilko-ES (South End spawners) and Chilko-S (all other spawners) cannot be disaggregated, so independent assessments of these two CUs were not possible. No gap filling was required for this system (Appendix 2). This system was enumerated using mark recapture methods up to 2008, with the exception of 1967, which was estimated based on the expansion of counts at Henry's Bridge. In 2009, DIDSON methods were used to estimate abundance. Chilko River (including the North End of Chilko Lake) comprises 98% of the total abundance in years when the South End (of the lake) spawners and channel were estimated separately.

Productivity: Similar to other Summer Run CUs, the Chilko-ES & Chilko-S CU aggregate has exhibited systematic declines in productivity (Kalman filter Ricker *a* parameter values) since the 1990 brood year (Appendix 3, Chilko-ES & Chilko-S, Figure 1 c). Productivity (R/S) has been particularly low recently (1994 to 2005 brood years), with six years close to or below replacement (Appendix 3, Chilko-ES & Chilko-S, Figure 1 d). Similar to other CUs with freshwater survival data, the Chilko-ES and Chilko-S aggregate survival (smolts/EFS) decreased consistently from the mid-1960 to 2000 brood years, and has subsequently increased (Appendix 3, Chilko-ES & Chilko-S, Figure 1 e). Record numbers of smolts were reported in the 2007 and 2008 smolt outmigration years (average: 75 million) relative to the time series average (average: 20 million). Marine survival (recruits/smolt) has decreased consistently from the 1990 to 2005 brood years (Appendix 3, Chilko-ES & Chilko-S, Figure 1 f).

Abundance: The full stock-recruitment time series for Chilko-ES & Chilko-S includes the brood years 1950-2004. For Ricker model benchmark estimates (recommended model by Holt et al. 2009), a lognormally distributed prior (mean of 400,000 and sigma of 16,000), based on calculations of lake rearing capacity, was used to estimate the Ricker '*b*' parameter (Table 4; Appendix 3, Chilko-ES & Chilko-S, Figure 2 d). Using the standard Ricker model and the full time series (brood years 1950-2004), lower benchmarks ranged from 28,000 to 54,000 at the 10% to 90% p-levels and upper benchmarks ranged from 238,000 to 311,000 (Table 4; Appendix 3, Chilko-ES & Chilko-S, Figure 2 e). Statuses were assessed by comparing both the arithmetic (275,000) and the geometric (248,700) mean abundance of the last generation to the benchmarks. The resulting statuses decreased at higher probability levels. Specifically, status changed from Green to Amber above the 50% and 10% p-levels, respectively, when evaluated using the arithmetic and geometric mean abundances (Table 4). Statuses were similar using these two calculations of recent abundance, as geometric and arithmetic recent generation means were similar (only a 10% difference between arithmetic and geometric means).

Given that the Chilko-ES & Chilko-S aggregate has exhibited systematic declines in productivity, model forms that specifically consider this recent lower productivity in benchmark estimation produced higher (more biologically conservative) lower benchmarks. The recursive-Bayesian Ricker model, followed by the smoothed-Ricker and most truncated time series (brood years 1990-2004) Ricker model produced the highest lower benchmarks. Upper benchmarks were lower for all models that specifically consider recent productivity, compared to the full time series standard Ricker model. Statuses for these model forms that consider recent productivity, decreased at higher probability levels; status changed from Green to Amber at p-levels greater than 50% for the arithmetic mean abundance and at p-levels greater than 25% for the geometric mean (Table 4).

Trends in Abundance: The Chilko-ES & Chilko-S CU exhibited a period of particularly high escapement (EFS) from 1990 to 2000 (average EFS: 400,000), relative to the time series average (192,000). Subsequently, this CU has declined from a period of above average EFS, which occurred three generations prior to the end of the time series (407,000), to the current generation average EFS (154,000) (Table 5; Appendix 3, Chilko-ES & Chilko-S, Figure 1 b). This CU exhibited strong cyclic dominance from 1950 to 1990 (dominant cycle average escapement: 250,000; one weak cycle average EFS: 39,000; and two subdominant cycles average EFS: 117,000;) (Appendix 3, Chilko-ES & Chilko-S, Figures 1 a & b). After 1990, cyclic dominance disappeared (Appendix 3, Chilko-ES & Chilko-S, Figures 1 a & b). Spawner success has remained generally high throughout the time series (~92%) and has not exhibited any persistent trends; with the exception of 2008 when spawner success was low (53%) (Appendix 3, Chilko-ES & Chilko-S, Figure 1 b).

The ratio of the recent generation geometric mean to the long-term geometric mean for Chilko-ES and Chilko-S EFS (ratio: 1.22) is greater than the upper benchmark for this metric (ratio: 0.75) (Green status) (Table 5; Appendix 2, Chilko-ES & Chilko-S, Figure 2 c). In recent years (last three generations), Chilko-ES & Chilko-S EFS has decreased, following a period of above average EFS (see previous paragraph). The slope of this recent trend (-79% change in abundance) is steeper than the lower benchmark for this metric (-25% change in abundance)(Red status), and there is a 100% probability that this recent trend is below the lower benchmark (Table 5; Appendix 3, Chilko-ES and Chilko-S, Figures 2 a & b).

Chilliwack-ES

Sites: Populations that rear in Chilliwack Lake include Chilliwack Lake and Dolly Varden Creek (also known as Upper Chilliwack River) (Appendix 1).

History: Chilliwack Lake is a relatively isolated lake surrounded by glaciated mountains. This lake is influenced by glacial melt and, therefore, given increasing temperatures and associated decreases in glacial mass, it has been warming. The Chilliwack-ES CU is amongst the first population of Sockeye to enter the Fraser River, with an entry timing more closely associated with the Early Stuart Sockeye than other Early Summer Run Sockeye. Chilliwack-ES Sockeye spawn in the lake and in Dolly Varden Creek from late August to early September.

Escapement Time Series: Chilliwack Lake assessments began in the 1970's but were only consistently assessed starting in 1982, with generally two or more visual (boat) surveys conducted annually. Carcass counts are expanded based on survey effort, using methods established during studies on the Taseko Lake population. The estimates are likely biased low given limitations in the number of carcasses that reach the lake surface after becoming moribund (Patterson *et al.* 2007b). Lake counts may be further compromised on survey days with heavy rain or winds, which decrease the visibility of carcasses on the lake surface.

Dolly Varden Creek has only been consistently assessed in more recent years, starting in 2001, and represents the bulk of the spawning (>70% of the total lake plus creek EFS) in the CU. Dolly Varden Creek is assessed using peak live and cumulative dead (helicopter) surveys. In 2001, a tower count was used to assess the total escapement to the lake and river combined, and a visual (helicopter) survey was conducted on Dolly Varden Creek; the lake was then estimated by subtracting the tower count from the creek estimate. Since the lake was also coincidentally assessed in 2001 using standard lake survey methods, both tower and visual survey escapement estimates were compared and no significant deviations occurred. No gap filling occurred for this CU as it did not have sufficient data for the analysis of trends in abundance (see proceeding *Trends in Abundance* section and Appendix 2).

Productivity: Productivity and survival could not be estimated for this CU as there are no associated recruitment data available for this CU.

Abundance: An alternative approach was used to estimate abundance benchmarks for this CU, given that there are no stock-recruitment data. Instead of using stock-recruitment models to estimate benchmarks, available S_{max} estimates from juvenile lake rearing capacity were used. The recommended lower benchmark and upper benchmark using this approach are respectively, 20% and 40% of spawners at maximum juvenile production (S_{max}) estimated from freshwater production studies (Holt et al. 2009). For Chilliwack Lake, the average number of spawners at maximum recruitment is 41,000 (Appendix 4 C), therefore, lower and upper benchmarks for Chilliwack-ES are, respectively, 8,000 and 16,000. The recent generation arithmetic mean abundance for Chilliwack-ES (12,000) was between the lower and upper benchmarks (Amber status) and the arithmetic mean abundance (5,000) was below the lower benchmark (Red status (Table 4). This metric could not be assessed probabilistically, therefore, only the deterministic lower and upper benchmarks and associated status are presented.

Trends in Abundance: Chilliwack Lake has exhibited variable escapement throughout the time series, and was particularly low in abundance in the last generation (average EFS: 500) relative to the long-term average (average EFS: 1,100) (Table 5; Appendix 3, Chilliwack-ES, Figure 1 b (Chilliwack Lake only)). Since Dolly Varden Creek assessments only commenced post-2000, in these years this creek has exhibited a considerably shallower declining trend compared to Chilliwack Lake. Dolly Varden Creek exhibited high EFS in three years (2001, 2004 & 2008 average EFS: 34,000) and weaker EFS in all other years assessed (average EFS: 2,000) (Appendix 3, Chilliwack-ES, Figure 1 a). In years when the Dolly Varden Creek population was large (2001, 2004 & 2008), it comprised 94% of this CU's total escapement. In weaker abundance years for Dolly Varden Creek, the creek comprised 54% of the total escapement for this CU. Given that Dolly Varden Creek comprises a greater average proportion of the total EFS (Dolly Varden Creek plus Chilliwack Lake post-2001) compared to Chilliwack Lake but only has been recently assessed with any degree of precision and accuracy (post-2000)(Appendix 2), and the trends in the EFS time series considerably differ between the two assessed sites, *trends in abundance* metrics could not be quantitatively assessed for this CU. It would be misleading to present *trends in abundance* statuses for Chilliwack Lake alone.

Cultus-L

Sites: Cultus Lake Sockeye are the only Sockeye population that rears in Cultus Lake (all spawners spawn in Cultus Lake) (Appendix 1).

History: Cultus-L has been the most intensively studied salmon CU in British Columbia. Studies on spawner abundance, lake characteristics and juvenile production began with the work of the Pacific Biological Station in the 1920's, and have continued into the present with the work of the International Pacific Salmon Fisheries Commission (IPSFC) and the Department of Fisheries and Oceans (DFO) (Schubert *et al.* 2003). Cultus-L Sockeye spawner abundance was low and variable during large scale hatchery experimentation in the 1920's and 1930's, very high in 1939-1942 following removal of predators, strong but variable in the early 1940's to late 1960's, and has subsequently declined. Exploitation rates were high from 1952 to 2002 (average: 67%), since this population co-migrates with more abundant and productive CUs (Harrison (U/S)-L, Shuswap Complex-L). Beginning in 1995, ER's decreased to an average of 33%. In 2001 and 2002, the Fraser River Panel (FRP) and DFO limited fisheries on Late Run populations to ER's of ~20% (Fisheries and Oceans Canada 2010; Bradford *et al.* 2010a). Three main causes for the decline of Cultus-L Sockeye include high exploitation rates between 1952 and 1995, high pre-spawn mortality (coincides with early migration of Late Run Sockeye starting in 1995), and low marine survival, particularly in recent years. Other causes may include heavy recreational, residential and agriculture land use around the lake, the loss of spawning habitat attributed to water milfoil invasion, and predation threats (Schubert *et al.* 2003; COSEWIC 2003; Cultus Sockeye Recovery Team 2009).

As a result of significant population declines in this CU, Cultus-L is listed as *Endangered* by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC)(25 October 2002)(COSEWIC 2003). Consequently, a Cultus Lake Sockeye Recovery Planning Team was formed in 2002, with both internal-DFO and non-DFO representation, to document status and develop a recovery plan (Cultus Sockeye Recovery Team 2009). This team was disbanded after the publication of the Cultus Recovery Strategy (Cultus Sockeye Recovery Team 2009), which outlined an overall conservation goal and four key objectives.

Subsequently, a Cultus Conservation team (similar DFO membership to the Recovery Team) was formed to continue with recovery work, and track recovery efforts and status. A recent publication (Research Document and corresponding Science Advisory Report), as part of the Canadian Science Advisory Secretariat (CSAS) process, has been published (Bradford et al. 2010; Fisheries and Oceans Canada 2010). Recovery actions to date have included hatchery enhancement (captive brood stock and hatchery supplementation programs), predator (Pikeminnow: *Ptychocheilus oregonensis*) removal, and harvest reductions. This Conservation Team publication concludes that although the decline in Cultus-L Sockeye has been halted, the population has not yet met any of the recovery objectives set by the Cultus Sockeye Recovery Team. The prospects for Cultus-L Sockeye are highly uncertain, and are tied to future trends in marine (recruits/smolt) survival which have been particularly low in recent years. Recovery actions in recent years have included reductions in harvest (~20% ER), predator control in Cultus Lake (which has coincided with an increase in in-lake survival of juvenile Sockeye salmon), and a captive broodstock/supplementation program (majority of adults returning in 2008 & 2009 were of hatchery origin).

Escapement Time Series: Only Cultus Lake was included in the escapement time series (Appendix 1). Cultus Lake Sockeye have been assessed since 1925 using an enumeration fence in Sweltzer Creek, located approximately 200 m downstream of the lake outlet. The fence is installed at the start of the migration period (normally mid/late September), and is removed at its completion in early/mid-December. As this CU started to migrate earlier in the mid-1990's, fence installation has occurred at progressively earlier dates, with installation in recent years occurring in August. There are no gaps in the time series (Appendix 2).

For the escapement time series, effective total adult escapement (total adult escapement multiplied by female spawner success) was used instead of effective female escapement for *trends in abundance* analyses, due to uncertainty in sex identification at the fence. Cultus Sockeye do not have well developed secondary sexual characteristics when assessed at the Cultus fence because they migrate through the fence early, and move into the deeper and cooler lake where they hold for months before spawning in December to January. Calculation of spawner success is typically based on the assessment of carcasses on the spawning grounds. However, given low abundances of Cultus-L Sockeye in recent years, recovery of female carcasses has been negligible. Therefore, a combination of spawner success data from the enhancement program (Cultus Sockeye captured at the fence and retained in holding ponds for hatchery purposes), Weaver Creek & Channel data, and data on Cultus-L Sockeye recruits-per-juveniles, was used to assess spawner success for Cultus Sockeye (Bradford et al. 2010). Post-2000, due to hatchery enhancement of this system (Schubert *et al.* 2003; Cultus Sockeye Recovery Team 2009; Bradford *et al.* 2010a), only wild unmarked fish (no adipose-fin clip) were included in the escapement time series.

Productivity: Similar to other CUs, Cultus-L has exhibited systematic declines in productivity (Kalman filter Ricker *a* parameter values) since the 1990 brood year (Appendix 3, Cultus-L, Figure 1 c). Productivity (R/S) has been particularly low recently (1993 - 2005 brood years), with seven of these years close to or below replacement (Appendix 3, Cultus-L, Figure 1 d). Freshwater and marine survival trends are a challenge to interpret, due to considerable gaps in the smolt, and therefore, survival time series (Figures 7 A & B; Appendix 3, Cultus-L, Figures 1 e & f). In years where it exists, the marine survival (recruits/smolt) time series tends to correspond to the Chilko survival time series (Figure 7 B).

Abundance: For Cultus-L Sockeye, only the brood years from 1950-2000 were used to estimate abundance benchmarks. Although brood years 2001 - 2003 have full recruitment data (age-4 plus age-5 recruits), these years were not included due to the confounding influence of the hatchery programs, which have contributed fry and smolts to Cultus Lake production and are unaccounted for in the spawner-recruit relationship. Although the hatchery programs started in 2000, the number of fry produced in the first year of operation was negligible, therefore stock-recruitment data for this year can be included in the time series. For Ricker model benchmark estimates (recommended model by Holt et al. 2009), a lognormally distributed prior (mean of 80,000 and sigma of 12,000), based on calculations of lake rearing capacity, was used to estimate the Ricker '*b*' parameter (Table 4; Appendix 3, Cultus-L, Figure 2 d). Using the standard Ricker model and the full time series (brood years 1950-2000), lower benchmarks ranged from 9,000 to 17,000 at the 10% to 90% p-levels, and upper benchmarks ranged from 28,000 to 36,000 (Table 4; Appendix 3, Cultus-L, Figure 2 e). Statuses were assessed by comparing both the arithmetic (900) and the geometric (600) mean abundance of the last generation to the benchmarks. The resulting statuses were Red across all p-levels (Table 4) given that abundances in the last four years (2006-2009) were low relative to all lower benchmarks (all model forms and probability levels).

Although Cultus-ES has exhibited systematic declines in productivity, model forms that specifically consider this recent lower productivity in benchmark estimation produced similar benchmarks to the full time series standard Ricker model. The smoothed Ricker model produced slightly higher lower benchmarks across all probability levels. Other model forms produced lower benchmarks that fall both above and below the full time-series Ricker model, depending on the probability level. Upper benchmarks were similar or lower for these model forms, compared to the full time series standard Ricker model. Statuses for model forms that consider recent productivity were also identical (Red status) across all p-levels.

Trends in Abundance: Cultus-L has experienced three distinct periods of abundance (Appendix 3, Cultus-L, Figures 1 a & b). The earliest years in the time series (1934 - 1968) exhibited the highest average effective total spawners (ETS) at 19,400, with peak escapements occurring from 1939 to 1942 (average ETS: 45,500), following predator removal from Cultus Lake. This early period of abundance was strong but variable with no cyclic dominance, attributed to the operation of the Sweltzer hatchery and periodic control of predators feeding on Sockeye fry in the lake (Cultus Sockeye Recovery Team 2009). Abundance subsequently declined during the period from 1960 to 1991 (average ETS was 8,200). During this period, cyclic dominance occurred, with three stronger cycles and one weaker cycle. In recent years (1992 to 2009), average ETS has declined further to 1,600, and cyclic dominance has again disappeared. Female spawner success was relatively high (92%) from 1934 to 1992. In recent years, spawner success has decreased (74%), with some years as low as 15% (e.g. 1999 and 2000) (Appendix 3, Cultus-L, Figure 1 b).

The ratio of the recent generation geometric mean 'wild (unclipped)' Sockeye abundance to the long-term geometric mean for Cultus-L (ratio: 0.07) is well below the lower benchmark for this metric (ratio: 0.5) (Red status) (Table 5; Appendix 3, Cultus-L, Figures 2 c). In recent years (last three generations), Cultus-L has declined in 'wild' Sockeye abundance following a period of already below average EFS (see previous paragraph). The slope of this recent trend (-73% change in abundance) is steeper than the lower benchmark for this metric (-25% change in abundance)(Red status), and there is a 100% probability that this recent trend is below the lower benchmark (Table 5; Appendix 3, Cultus-L, Figures 2 a & b).

Francois-Fraser-S

Sites: There are three sites for this CU, the Stellako River and Uncha and Ormonde Creeks (Appendix 1).

History: After the 1913 Hells Gate landslide, the Francois-Fraser-S population began to build and exhibit cyclic dominance. In 1964, log driving commenced on the Stellako River, to transport logs downstream from upriver forestry operations. This practice moved logs by releasing large volumes of water from splash dams during the spring freshets. Log driving degraded the river system, leaving bark and wood fibre deposits on the river bottom and spawning grounds, and eroding river banks through scouring and log jams (Roos 1991). After 1968, log driving was discontinued. This CU has not exhibited cyclic dominance since the log driving period (Schubert 2000).

Significant hydro-electric infrastructure exists on the Nechako River, which is connected to the Stellako system. However, Francois-Fraser-S Sockeye habitat has not been affected, as the dam was constructed upstream of Sockeye accessible areas. Although flow management associated with this facility has likely historically affected Sockeye, current flows are managed to meet temperature targets for this species.

Escapement Time Series: Only the Stellako River was included in the escapement time series and no gap filling was required (Appendices 1 & 2). Escapement enumeration included mark recapture programs from 1950 to 1993 and from 2007 to 2009, and a fence program from 1994 to 2006. In 1994 and 1995, both mark recapture and fence counts were conducted to evaluate mark recapture biases (Schubert 2007); fence data were used as the escapement time series for these years. The comparison study concluded that sampling biases in the mark recapture program were bi-directional, and, as a result, were cumulatively small (Schubert 2007). Most Stellako River Sockeye (Summer Run timing) rear in Fraser Lake, with a smaller proportion of juveniles (from spawners that spawn near the top end of Stellako River at the outlet of Francois Lake) that rear in Francois Lake. Mark-recapture tagging data confirm the mixing of all temporal components spatially throughout the Stellako River; although the earliest arrivals tend to head to the top of the Stellako River and hold at the lake outlet before dropping back to spawn, these earliest arrivals spawn throughout the entire river. Similarly, later arrivals also spawn throughout the river including the uppermost reaches. Uncha Creek was not included in the escapement time series, as Sockeye in this creek have only been sporadically observed; low water frequently presents a barrier to Sockeye in many years. Uncha Creek Sockeye are Stellako River Sockeye strays that are observed only intermittently when Stellako abundances are high and when water levels permit entry of these fish into this creek. Ormonde Creek was also excluded from the escapement time series. These Sockeye historically spawned earlier (Early Summer timing), compared to recent years where spawning occurs later (Summer Run timing). This historical Early Summer timed Ormonde Creek Sockeye population is now considered extirpated (see proceeding extirpated section: Fraser-ES). In recent years, Sockeye in this creek are now assumed to be Stellako strays as they are observed particularly on more abundant Stellako years.

Productivity: Similar to other Summer Run CUs, Francois-Fraser-S has exhibited systematic declines in productivity (Kalman filter Ricker *a* parameter values) since the 1990 brood year (Appendix 3, Francois-Fraser-S, Figure 1 c). Productivity (R/S) has been particularly low recently (1998 - 2005 brood years), with six of these years close to or below replacement (Appendix 3, Francois-Fraser-S, Figure 1 d). Similar to other CUs with freshwater survival data, Francois-Fraser-S early freshwater survival (fry/EFS) decreased from the 1990's to the 2000 brood year, and has subsequently increased in the last year of this time series (fry were assessed in this system only from 1990 to 2002 brood years) (Appendix 3, Francois-Fraser-S, Figure 1 e). Post-fry survival (recruits/fry), that includes a period of freshwater survival and marine survival, generally decreased (Appendix 3, Francois-Fraser-S, Figure 1 f).

Abundance: The stock-recruitment time series for Francois-Fraser-S includes brood years 1950-2004. For Ricker model benchmark estimates (recommended model by Holt et al. 2009), a lognormally distributed prior (mean of 550,000 and sigma of 13,000), based on calculations of lake rearing capacity, was used to estimate the Ricker '*b*' parameter (Table 4; Appendix 3, Francois-Fraser-S, Figure 2 d). Using the standard Ricker model and the full time series (brood years 1950-2004), lower benchmarks ranged from 27,000 to 68,000 at the 10% to 90% p-level, and upper benchmarks ranged from 151,000 to 264,000 (Table 4; Appendix 3, Francois-Fraser-S, Figure 2 e). Statuses were assessed by comparing both the arithmetic (87,500) and the geometric (68,000) mean abundance of the last generation to the benchmarks. The resulting statuses were Amber across all probability levels (Table 4). Statuses were similar using these two calculations of recent abundance, as geometric and arithmetic recent generation means were similar (only a 22% difference between arithmetic versus geometric means).

Given that Francois-Fraser-S has exhibited systematic declines in productivity, model forms that specifically consider this recent lower productivity in benchmark estimation produced higher (more biologically conservative) lower benchmarks. The recursive-Bayesian Ricker model produced the highest (most biologically conservative) lower benchmarks, followed by the most truncated (brood years 1990-2004) Ricker model and the smoothed-Ricker. The less truncated Ricker model (brood years 1970-2004) produced similar lower benchmarks to the full time series standard Ricker model. Upper benchmarks were similar for the truncated Ricker models, compared to the full time series standard Ricker model, and lower for the recursive-Bayesian Ricker and the smoothed-Ricker models. Statuses for these model forms that consider recent productivity were Amber (or Green in the case of the smoothed-Ricker) up to the 25% p-level and then changed to Red at higher probability levels (exact level differs between models)(Table 4). Only the smoothed-Ricker model had Green statuses in the probability distribution (specifically at the 10% to 25% p-levels).

Trends in Abundance: The average abundance in the Francois-Fraser-S CU was low in the first half (1950-1974) of the time series (average: 32,300 EFS) and increased from 1975 to 2002 (average EFS: 70,800), with increasing frequency of high abundance years (exceeding 150,000 EFS). Average EFS across the entire time series is 53,000 (Table 5). This CU has declined from a period of above average EFS, which occurred three generations prior to the end of the time series (105,000), to the current generation average EFS (47,300) (Table 5; Appendix 3, Francois-Fraser-S, Figure 1 b). From 1950 to 1968, Francois-Fraser-S exhibited cyclic dominance, with one dominant cycle (average EFS: 61,500), one subdominant cycle (average EFS: 41,200) and two off cycles (average EFS: 19,700) (Appendix 3, Francois-Fraser-S, Figures 1 a & b). After this period, abundance fluctuated, with no persistence of cyclic dominance but large inter-annual variability in abundance. Throughout the time series, spawner success has remained high (~90%) and has not exhibited any persistent trends (Appendix 3, Francois-Fraser-S, Figure 1 b).

The ratio of the recent generation average abundance to the long-term average (ratio: 1.31) for Francois-Fraser-S is above the upper benchmark (Green status) (ratio: 0.75) (Table 5; Appendix 3, Francois-Fraser-S, Figure 2 c). In recent years (last three generations), Francois-Fraser-S EFS has decreased following a period of above average EFS (see previous paragraph). The slope of this recent trend (-38% change in abundance) is steeper than the lower benchmark for this metric (-25% change in abundance)(Red status), and there is a 78% probability that this recent trend is below the lower benchmark (Table 5; Appendix 3, Francois-S, Figures 2 a & b).

Harrison (D/S)-L

Sites: Populations that migrate downstream to rear in Harrison Lake, after emerging from the gravel as fry, include Bear Creek, Big Silver Creek, Cogburn Creek, Crazy Creek, Douglas Creek, Hatchery Creek, Sloquet Creek, Tipella Creek and Tipella Slough (Appendix 1).

History: Big Silver Creek, the most consistently assessed stream in this CU, originates in the Lillooet Range of the Coast Mountains east of Harrison Lake, and flows predominantly west to the lake. River flows are maintained throughout the summer via snowfields in the headwaters. Although the total length of the Big Silver mainstem, from headwaters to mouth, is approximately 40 km, a waterfall 6 km from the mouth prevents fish passage further upstream. The lower 15 km of the mainstem channel is very stable and contains only a single major bifurcation 2 km up from the mouth. Big Silver contains numerous narrow bedrock canyons spread sporadically through the length of the mainstem. Stream banks are stable and serve to confine the river during periods of high flow (Wilson et al. 1999). Big Silver was historically affected by logging activities, which may have changed flow regimes, sediment deposition, and caused erosion (Fisheries and Oceans Canada 1999). Restoration and enhancement projects have been conducted on Big Silver Creek, aimed specifically at enhancing flows and Sockeye usage of the north fork of this creek where high quality spawning habitat (classic spawning gravel), relative to the south fork (large cobbles), occurs (K. Peters, DFO, pers. comm.).

Escapement Time Series: Big Silver Creek is the only creek consistently assessed in this CU, likely due to ease of surveyor accessibility (Appendix 1). Douglas, Hatchery and Bear Creeks were assessed in 1950-1953 (in these year's, Big Silver Creek comprised 50% of the total escapement) and Cogburn, Crazy, Sloquet and Tippella Creeks were assessed only after 2000 (in these year's, Big Silver comprises 92% of the total escapement). Therefore, only Big Silver Creek is included in the escapement time series and no gap filling was required (Appendix 2).

Productivity: Productivity and survival could not be estimated for this CU as there are no associated recruitment data available for this CU.

Abundance: Abundance benchmarks could not be estimated for this CU as there are no associated recruitment data available. Information on spawning capacity can also not be used since this CU rears in Harrison Lake along with Harrison (U/S)-L that makes up a more significant proportion of total fry that rear in this lake. Spawning habitat capacity likely limits juvenile production rather than lake rearing capacity.

Trends in Abundance: Harrison (D/S)-L is a small CU with an average EFS of 1,500 (Appendix 3, Harrison (D/S)-L, Figure 1 b). From 1964 to 1998, abundance was relatively low (average EFS: 580), and subsequently increased between 1999 and 2009 (average EFS: 5,400) (Table 5). From 1950 to 1964, Harrison (D/S)-L Sockeye exhibited cyclic dominance, with one dominant (average EFS: 2,500) and three subdominant cycles (average EFS: 100). Since 1964, this CU has not exhibited cyclic dominance. Throughout the time series, spawner success has remained high (~85%), with a few intermittent years of low spawner success (1953: 30%; 1981: 67%; 1983: 54% and 2008: 63%) (Appendix 3, Harrison (D/S)-L, Figure 1 b).

The ratio of the recent generation average abundance to the long-term average (ratio: 13.3) for Harrison (D/S)-L is well above the upper benchmark (Green status) (ratio: 0.75) (Table 5; Appendix 3, Harrison (D/S)-L, Figure 2 c). This CU's abundance has increased in the last three generations with a positive slope (274%) that is well above the upper benchmark for this metric (15% rate of decline) (Green status), and there is an extremely small probability (1%) that this recent increasing trend is below the lower benchmark for this metric.

Harrison (U/S)-L

Sites: Populations that migrate upstream to rear in Harrison Lake, after emerging from the gravel as fry, include East Creek (rolled up into Weaver Creek after 1951 and may alternatively be named Sakwi Creek), Steelhead Creek (rolled up into Weaver Creek throughout the time series), Weaver Creek and Weaver Channel (Appendix 1).

History: Until 1965, Weaver Creek was the key producer of Sockeye in this CU (average EFS: 9,200), with negligible contributions from Steelhead Creek (small creek on the west side of Weaver Creek near a swampy area) and East Creek (located on the east side of Weaver Creek). Towards the end of this early period (1961-1964), extensive logging within the watershed caused considerable flooding and scouring of Sockeye spawning habitat, and abundance declined to near extinction (Roos 1991). Substantial erosion and sediment input into Sakwi and Weaver Creeks occurred as a result of logging (1963), and road and trail clearing associated with the development of a ski resort (1970's) (Rood and Hamilton 1995). In the 1972 brood year, a decline in fry/egg survival in Weaver Creek was attributed to the accumulation of sediment and organic debris in the gravel. Gravel cleaning returned survival to normal by the 1973 brood year, but it declined again in 1974 to 1975 for the same reason (International Pacific Salmon Fisheries Commission 1972).

The Weaver Creek diversion weir and spawning channel (located on Weaver Creek, upstream of Harrison River), the first of its kind for Sockeye in BC, was built in the mid-1960's, and started operating in the fall of 1965. Weaver channel was constructed to re-build production from the Weaver stock, and subsequently allow for increased harvest opportunities on the Late Run CUs (which includes the large Adams River run). The channel also serves to protect the Weaver run from periodic flooding events. A flow control structure is operated at the outlet of Weaver Lake, to manage the water supply for channel operations. Sakwi Creek, a tributary of Weaver Creek, upstream of the channel, also has an intake that provides water for the channel as required.

The channel operated at 25% of capacity until 1969, when there were sufficient spawners to fill it to near capacity (International Pacific Salmon Fisheries Commission 1972). Subsequently, Sockeye were preferentially diverted into the channel over the creek, since their presence in the creek is thought to affect oxygen concentrations in the channel's source water. The channel has approximately eight times higher fry/egg survival compared to the creek (natural spawning grounds), based on data available from 1965-1988. Pre-spawn mortality has been relatively high in the last four years, attributed to the *Parvicapsula* parasite. The cause of *Parvicapsula* outbreaks is not yet clear, although it is thought to be associated with changes in river entry timing and water temperatures (R. Cook, DFO, pers. comm.). There has also been one year (1995) of elevated pre-spawn mortality associated with *Ichthyophthirius multifiliis*. Although this pathogen typically does not cause disease in Sockeye Salmon, "ich" or "white spot disease" can occur if numbers of this pathogen are high due to conditions such as warm water, reduced flows, and adult crowding.

Weaver has historically had low flow levels and was essentially dry during the 1952 drought (Rood and Hamilton 1995). Channel excavation is conducted annually in lower Weaver Creek to maintain a low flow channel and holding pools, to improve conditions for salmon migration during low flow conditions. Weaver has also been dredged a number of times to maintain access to the spawning channel (Rood and Hamilton 1995).

Escapement Time Series: Three sites are included in the escapement time series: Weaver Creek, Weaver Channel and East Creek and no gap filling was required (Appendices 1 & 2). East Creek has independent data early in the time series but was included in the Weaver Creek estimate after 1951. Steelhead Creek is not included separately in the escapement records, and has also been rolled up into the totals for Weaver Creek. Weaver Creek and Channel cannot be evaluated independently, since numbers of Sockeye in each is a consequence of channel loading regimes at the diversion fence located at the outlet of this system. Data for the channel begins in 1965 after its construction. From 1950 to 1988, mark recapture surveys were primarily used to assess escapement into Weaver Creek (with the exceptions of 1951, 1966-1968, which were assessed with peak live cumulative dead methods). From 1989 to 2009, peak live cumulative dead visual surveys were conducted (with the exceptions of 1994, 1996 and 1998, which were assessed using mark recapture methods, and from 1999-2000 and 2002-2003, which were assessed using an enumeration fence). Weaver Channel was exclusively assessed at the channel diversion fence, using counts of live Sockeye migrating above the diversion weir to the spawning channel, the upper creek, and into the ESSR holding channel. Fish removed for ESSR were not counted in escapements. Visual surveys were conducted in lower Weaver Creek. downstream of the diversion fence, and carcass surveys were conducted upstream and downstream of the diversion fence.

Productivity: In contrast to Early Summer Run and Early Stuart CUs, the Harrison (U/S)-L CU has not exhibited any persistent trends in productivity (based on Kalman filter Ricker *a* parameter values)(Appendix 3, Harrison (U/S)-L, Figure 1 c). However, productivity (R/S) has been particularly low in recent years (2000 to 2005 brood years), with one of these years falling below replacement (Appendix 3, Harrison (U/S)-L, Figure 1 d). Similar to other CUs with freshwater survival data, Harrison (U/S)-L early freshwater survival (fry/EFS) decreased consistently from the start of the time series in 1966 up to 1990, and has subsequently increased (Appendix 3, Harrison (U/S)-L, Figure 1 e). Post-fry survival (recruits/fry), which includes a period of freshwater survival and marine survival, decreased in the 1970's, increased in the 1990's and has subsequently decreased (Appendix 3, Harrison (U/S)-L, Figure 1 f).

Abundance: The Harrison (U/S)-L stock-recruitment time series only includes years after the construction of the Weaver spawning channel (brood years 1966-2004), to ensure consistency in the spawning area throughout the time series. For Ricker model benchmark estimates (recommended model by Holt et al. 2009), a uniformly distributed prior (range: 0 to 1,000,000) was used to estimate the Ricker '*b*' parameter (Table 4; Appendix 3, Harrison (U/S)-L, Figure 2 d). Using the standard Ricker model and the full time series (brood years 1966-2004), lower benchmarks ranged from 4,000 to 23,000 at the 10% to 90% p-level and upper benchmarks ranged from 52,000 to 147,000 (Table 4; Appendix 3, Harrison (U/S)-L, Figure 2 e). Statuses were assessed by comparing both the arithmetic (20,400) and the geometric (12,200) mean abundance of the last generation to the benchmarks. The resulting statuses decreased at higher probability levels. Specifically, status changed from Amber to Red above the 75% and 50% p-levels, respectively, when evaluated using the arithmetic and geometric mean abundances (Table 4).

Harrison (U/S)-L has not exhibited the systematic declines in productivity exhibited by other CUs. Therefore, model forms that specifically consider recent productivity in benchmark estimation did not produce consistently larger (more biologically conservative) lower benchmarks than the full time series standard Ricker model. The recursive-Bayesian Ricker benchmarks are not recommended for this CU, as they produce S_{max} estimates that are unrealistically high relative to other models. The smoothed-Ricker produced the highest lower benchmarks, and the most truncated time series (brood years 1990-2004) Ricker model produced the lowest, relative to the full time series standard Ricker model. Upper benchmarks were lower for these models, compared to the full time series standard Ricker model. Statuses for these model forms that consider recent productivity were generally Amber in status, with the exception of some of the highest probability levels, which were Red in status, depending on the model and method for estimating recent abundance (Table 4).

Trends in Abundance: Abundance in Harrison (U/S)-L was particularly low at the start of the time series, prior to channel construction (1950-1974 average EFS: 11,000), increased from 1975 to 1990 (average EFS: 32,500) and has decreased again in recent years (1990-2009 average EFS: 18,700). The average EFS for the entire time series is 19,200 (Table 5). Harrison (U/S)-L has not exhibited cyclic dominance throughout the time series (Appendix 3, Harrison (U/S)-L, Figure 1 a). Spawner success was consistently high from 1964 to 1994 (channel: 96%; creek: 95%) and lower from 1995 to 2009 (channel: 83%; creek: 57%). In the channel, the lowest spawner success years were 1995, 2006 and 2008 (~70% in each year). The creek had a large number of years with extremely low spawner success (2001: 8%; 2006: 14% and 2008: 7%, and many years after 1994 where success was 40-50% i.e. 1995-1997; 1999; 2009) (Appendix 3, Harrison (U/S)-L, Figure 1 b).

The ratio of the recent generation geometric mean to the long-term geometric mean for Harrison (U/S)-L EFS (ratio: 0.8) is greater than the upper benchmark for this metric (ratio: 0.75) (Green status) (Table 5; Appendix 2, Harrison (U/S)-L, Figure 2 c). For comparison, if only data after the installation of Weaver channel (1965-2009) are used to estimate the trend in abundance, the ratio of the recent generation abundance to the long-term average (0.62) would fall between the lower (ratio: 0.5) and upper (0.75) benchmark for this metric (Amber status). In recent years (last three generations), Harrison (U/S)-L EFS has decreased with a slope that is steeper (-30% change in abundance) than the lower benchmark for this metric (-25% change in abundance)(Red status), and there is a 60% probability that this recent trend is below the lower benchmark (Table 5; Appendix 3, Harrison (D/S)-L, Figures 2 a & b).

Harrison River (River-Type)

Sites: This CU includes a single site, the Harrison River (Appendix 1).

History: The only site with a consistent time series and a confirmed established river-type population in the Lower Fraser Area is the Harrison River (River-Type) Sockeye. The Harrison River system originates in the Coast Mountains and drains Harrison Lake. The mouth of the Harrison River forms a floodplain marsh approximately 0.05 km^2 in size. The Harrison Rapids at the outlet of the Chehalis River provide an important control on water levels at low discharge (Rood and Hamilton 1995). As a result, the Harrison River is very stable with coarse substrate. During the spring the rapids are backwatered and inundated by the freshet flows of the Fraser River (Fisheries and Oceans Canada 1999). The rapids and lower portion of the river, which are used by Sockeye for spawning habitat, have been dredged to maintain a navigation channel (Rood and Hamilton 1995). At higher discharges the river spreads to cover the main channel as well as three others where fish spawn (primarily pink spawning ground) (International Pacific Salmon Fisheries Commission 1972).

Harrison Sockeye are unique compared to other Fraser Sockeye CUs in terms of their freshwater residence, age structure, ocean migration timing, and migration routes. After Harrison Sockeye emerge from the gravel they are thought to rear in sloughs for a few months prior to their downstream migration, and, as a result, enter the Strait of Georgia a few months after all other Fraser Sockeye (Birtwell *et al.* 1987). Unlike other Fraser Sockeye, they do not rear in freshwater lakes as juveniles for one to two years. Also unlike all other Fraser Sockeye, Harrison Sockeye rear in the Strait of Georgia for up to six months, prior to migrating through the Southern Juan de Fuca Strait (Taylor *et al.* 1996; Tucker *et al.* 2009, R. Beamish, DFO, pers. comm.). All other Fraser Sockeye immediately migrate north through the Johnstone Strait once they reach the Strait of Georgia.

Escapement Time Series: The only site in the Harrison River (River-Type) CU is the Harrison River and no gap filling was required of this escapement time series (Appendices 1 & 2). Mark recapture programs were conducted on this system until 1971, and in 1978 to 1979. After 1971, peak live cumulative dead visual survey methods were typically used, largely via boat, and then via helicopter starting in 1994. Escapements increased dramatically beginning in 2005 (400,000 total adults), although it was not until 2009 that a mark recapture program was re-instituted. Escapement estimates between 2005 and 2008 are likely biased low due to the assessment challenges of visually counting large numbers of Sockeye. Visual assessments were compromised in four additional years (1986, 1989, 1991 and 1993), due to poor visibility in the lake. Overall, the use of visual surveys on the Harrison River introduces large negative biases, because observations are confounded by the size and depth of the river, and the large spawning populations of Chinook and Chum (Schubert 2007)

Productivity: Harrison (River-Type) Sockeye, unlike most other CUs, have increased in productivity in recent years, with the exception of the 2005 brood year, which had the lowest productivity on record for this CU (Appendix 3, Harrison River (River-Type), Figures 1 c & d). Mechanisms explaining the recent dramatic increase in productivity and abundance are poorly understood, although linkages to improved survival are likely related to their different life-history strategy (river-type) relative to most other Fraser Sockeye (lake-type).

Abundance: The full stock-recruitment time series for Harrison River (River-Type) includes the brood years 1950-2005. Since this CU is dominated by Harrison Sockeye, which return as three and four year old fish (rather than four and five year olds for all other Sockeye CUs), total recruitment data are available up to 2005 (only available to 2004 for all other Fraser Sockeye CUs). For Ricker model benchmark estimates (recommended model by Holt et al. 2009), a uniformly distributed prior (range: 0 to 800,000) was used to estimate the Ricker '*b*' parameter (Table 4; Appendix 3, Harrison River (River-Type), Figure 2 d). Using the standard Ricker model and the full time series (brood years 1950-2005), lower benchmarks ranged from 6,000 to 14,000 at the 10% to 90% p-levels, and upper benchmarks ranged from 28,000 to 40,000 (Table 4; Appendix 3, Harrison River (River-Type), Figure 2 e). Statuses were assessed by comparing both the arithmetic (147,700) and the geometric (80,300) mean abundance of the last generation to these benchmarks. The resulting statuses were Green across all probability levels (Table 4).

Given that the Harrison River (River-Type) CU has exhibited systematic increases in productivity, in contrast to most other CUs, which have exhibited decreases in productivity, model forms that specifically consider recent productivity in benchmark estimation generally produced smaller (less biologically conservative) lower benchmarks for this CU. The recursive-Bayesian Ricker benchmarks are not recommended for this CU as they produce S_{max} estimates that are unrealistically high relative to other models. The smoothed-Ricker model produced extremely small lower benchmarks relative to all other models. The less truncated (brood years 1970-2005) Ricker model produced similar benchmarks to the full time series (brood years 1950-2005) standard Ricker model, and the more truncated (brood years 1990-2005) Ricker model produced slightly smaller lower benchmarks. Upper benchmarks were generally higher for all models that specifically consider recent productivity, with the exception of the smoothed-Ricker model benchmarks, which were approximately one third the size of the full time series (1950-2005) standard Ricker model benchmarks. Statuses for these model forms that consider recent productivity were Green for all models and probability levels (Table 4).

Trends in Abundance: From 1950 to 2004, the Harrison River (River-Type) Sockeye CU was relatively small in terms of abundance (average EFS: 6,400) (Table 4; Appendix 3, Harrison River (River-Type), Figure 1 a). After 1994, abundance dramatically increased to a maximum of 200,000 EFS in 2005 (average EFS: 93,000). With the exception of one brood year in the recent time period, which experienced the lowest productivity on record for this CU (2005 brood year), this CU has been extremely productive and abundant. This CU has not exhibited cyclic dominance. Spawner success has also been consistently high throughout the time series (average: 98%) (Table 4; Appendix 3, Harrison River (River-Type), Figure 1 b).

The ratio of the recent generation geometric mean to the long-term geometric mean for Harrison (River-Type) EFS (ratio: 6.98) is well above the upper benchmark for this metric (ratio: 0.75) (Green status) (Table 5; Appendix 3, Harrison River (River-Type), Figure 2 c). In recent years (last three generations), Harrison River (River-Type) EFS has increased (see previous paragraph). The positive slope of this recent trend (2453% change in abundance) is well above the lower benchmark for this metric (-25% change in abundance)(Green status), and there is a 0% probability that this recent trend is below the lower benchmark (Table 5; Appendix 3, Harrison River (River-Type), Figures 2 a & b). Additionally, given that escapement estimates for recent years are likely negatively biased (underestimates) and imprecise (highly uncertain), this increasing trend is likely larger than the current trend analysis indicates.

Kamloops-ES

Sites: Populations that rear in Kamloops Lake include Barriere River, Clearwater River, Dunn Creek, Finn Creek, Grouse Creek, Hemp Creek, Lemieux Creek, Lion Creek, Mann Creek, Moul Creek, North Thompson and Raft Rivers (Appendix 1). A nearby system that includes Sockeye spawning populations in Fennell and Harper Creeks was not included in the Kamloops-ES CU, since these populations rear as juveniles in Barriere Lake and not Kamloops Lake. The Fennell and Harper populations are now included in both an extirpated North Barriere-ES CU (extirpated after construction of a downstream dam in this system) and a *de novo* North Barriere-ES *(de novo)* 'CU' (that originates from hatchery transplants after the downstream dam was removed in 1952), found in proceeding sections.

History: The only history found for this CU was that Raft Creek was used as a donor population for hatchery transplants into Fennell Creek, to re-build Fennell Creek after it was effectively extirpated due to a dam on Fennell Creek up to 1952 (Beacham *et al.* 2004).

Escapement Time Series: Only Raft River was included in the escapement time series and no gap filling was required for this site (Appendix 2). Raft River has been consistently assessed since 1950 since this system is relatively small and easy to access. Raft has been assessed using a combination of mark recapture and visual survey methods, with mark recaptures generally conducted during years of larger abundance. For most of the time series (1950-1999), Raft makes up ~90% of the total escapement. From 2000-2009, Raft's contribution to total escapement decreases to 60%, while the North Thompson River began to contribute larger escapements to the CU (roughly 40%). Despite its larger contribution to the CU abundances later in the time series, North Thompson was not included in the escapement time series for a number of reasons, including a shift in assessment methods that occurred in very recent years. Historically, the North Thompson was assessed using visual (ground) survey methods (peak live cumulative dead). This assessment method is particularly challenging for the North Thompson River because it is a large, extremely turbid system. Also, surveys generally occurred in the first week of September and, therefore, likely missed the peak of spawning. Starting in 2000, surveys were conducted by air during the third week of September. These more recent surveys likely better reflect true abundance in the system compared to previous assessments. During the methodology switch, abundance increased from an historical average of 400 EFS (prior to 2000) to a recent average of 164,000 EFS (2000-2009). The shift in assessment methods, as well as the size and turbidity of the system, confounds the ability to determine if the increase in abundance in the North Thompson River reflects actual trends, or is an artefact of methodology. It is likely that the change in abundance indicates a true increase, given that observations of large numbers of carcasses were not previously reported by DFO field assessment staff. Trends in the North Thompson also somewhat align with those of Raft, increasing in abundance starting in the late 1990's with a peak in escapement in 2005. Therefore, due to uncertainty in the North Thompson time series prior to 2000, and given similarities to Raft, North Thompson was not included in the trend analysis. Barriere River was also not included in the CU escapement time series as it has considerable gaps, negligible spawner abundance, and potentially poor quality data, due to the structure of the spawning substrate (big boulders), which makes visual ground surveys problematic. It is unclear whether these counts represent actual Barriere spawners or fish migrating through to Fennell and Harper Creeks. Therefore, Barriere was also not included in the escapement time series. All other enumeration sites were excluded from the escapement time series as they had negligible abundances that only started being observed near the end of the time series.

Productivity: Unlike other Early Summer Run and Early Stuart CUs, Kamloops-ES exhibited no changes in productivity (Kalman filter Ricker *a* parameter values) throughout the time series (Appendix 3, Kamloops-ES, Figure 1 c). However, in recent years, productivity (R/S) was particularly low from the 2003 to 2005 brood years, with two of these years below replacement (Appendix 3, Kamloops-ES, Figure 1 d). There are no freshwater or marine survival data available for this CU.

Abundance: The stock-recruitment time series for Kamloops-ES includes the years 1950-2004. For Ricker model benchmark estimates (recommended model by Holt et al. 2009), a uniformly distributed prior (range: 0 to 500,000) was used to estimate the Ricker '*b*' parameter (Table 4; Appendix 3, Kamloops-ES, Figure 2 d). Using the standard Ricker model and the full time series (brood years 1967-2004), lower benchmarks ranged from 3,000 to 15,000 at the 10% to 90% p-levels and upper benchmarks ranged from 15,000 to 50,000 (Table 4; Appendix 3, Kamloops-ES, Figure 2 e). Statuses were assessed by comparing both the arithmetic (9,500) and the geometric (9,000) mean abundance of the last generation to the benchmarks. The resulting statuses were Amber across all probability levels (Table 4), except for the 90% p-levels, which were Red. Statuses were identical using these two calculations of recent abundance, as geometric and arithmetic recent generation means were similar (only a 5% difference between arithmetic versus geometric means).

Although Kamloops-ES has not exhibited systematic declines in productivity, this CU has exhibited lower productivity in recent years. The smoothed Ricker models were not used to estimate benchmarks, as the removal of Fennell Creek from the escapement time series for the Kamlooops-ES CU occurred just prior to paper submission, and these additional analyses could not be conducted. The recursive-Bayesian Ricker model produced the highest lower benchmarks, while truncated Ricker models produced benchmarks that were similar to the full time series Ricker model; specifically, the least truncated (brood years 1970-2004) Ricker model produced only slightly higher (more biologically conservative) lower benchmarks compared to the standard Ricker model. The most truncated (brood years 1990-2004) Ricker model produced smaller (less biologically conservative) lower benchmarks compared to the full time-series (brood years 1950-2004) standard Ricker model. Upper benchmarks were similar for all models, with the exception of the recursive-Bayesian Ricker model that produced upper benchmarks that were three times greater than all other models forms. Statuses for the truncated Ricker models were generally Amber, except for at higher p-levels (at and above the 50% to 90% p-levels, depending on the model) that were Red. Statuses were identical when assessed using either the arithmetic or geometric recent mean abundances.

Trends in Abundance: Raft River EFS exhibited relatively low abundances from 1950 to 1990 (average EFS: 2,700), and started to build in the mid-1990's (average EFS from 1990-2009: 6,900). Average escapement across the entire time series (1950-2009) is 4,200 EFS (Table 5; Appendix 3, Kamloops-ES, Figure 1 b). Spawner success was generally high throughout the time series (average: 88%) with the exception of a few years (1950, 1967, 1971 and 1977) that were relatively low (~50%) (Appendix 3, Kamloops-ES, Figure 1 b).

The ratio of the recent generation geometric mean to the long-term geometric mean for Kamloops-ES (ratio: 2.37) is greater than the upper benchmark for this metric (ratio: 0.75) (Green status) (Table 5; Appendix 2, Kamloops-ES, Figure 2 c). In recent years (last three generations), Kamloops-ES EFS has increased (see previous paragraph). The slope of this recent increasing trend (16% change in abundance) is greater than the upper benchmark (-15% change in abundance) for this metric (Green status), and there is only a 12% probability that this trend is below the lower benchmark (Table 5; Appendix 3, Kamloops-ES Figures 2 a & b).

Lillooet-Harrison-L

Sites: Populations that rear in Lillooet and Harrison Lakes include the Birkenhead River, Green River, Lillooet Slough, Miller Creek, Poole Creek, Railroad Creek, Ryan River, Sampson Creek, John Sandy Creek, and Twenty-Five Mile Creek (Appendix 1).

History: Sockeye in the Lillooet-Harrison CU rear in both Lillooet Lake and Harrison Lake (Cave 1988). This CU is situated below the Fraser Canyon, and was not directly impacted by the 1913 Hells Gate landslide. Most spawning for this CU occurs in the Birkenhead River. Between 1946 and 1951, the course of the Birkenhead River was manually changed to flow directly into Lillooet-Harrison Lake (instead of via the Lillooet River) for the purpose of flood control (Rood and Hamilton 1995). This alteration likely reduced the potential spawning area (Schubert and Tadey 1997). Sections of the Birkenhead River and much of the lower 40 km of the upper Lillooet River have been dyked, and much of the floodplain has been ditched or filled, which has degraded salmon habitat. Changes to the system include wider shallower river channels with steeper gradients, channel degradation in the lower 13 km of Lillooet River, the isolation of cut-off meanders, a loss of wetlands, and a rapid increase in the rate of advance of the river delta (Schubert and Tadey 1997). In August 2010, a major landslide, caused by the Capricorn Mountain and Glacier giving way, resulted in rock and debris flows that blocked Meager Creek, located north of Pemberton. In 2010, returns of Sockeye will have to swim through a 1.5 km long suspended sediment wedge, as they enter the Birkenhead River. Although the remaining component of the Birkenhead River is not turbid, it is uncertain what impacts the suspended sediments will have on this population, and for how many years this will persist.

Escapement Time Series: Only the Birkenhead River was included in the escapement time series (Appendix 1). The Birkenhead River has been consistently assessed throughout the time period, and makes up over 99% of the escapement in years when other populations were also assessed. All other populations comprise only a minor component of total production for the Lillooet-Harrison-L CU, and these populations have only been opportunistically assessed with lower precision methods (visual ground surveys). Birkenhead River was assessed with a mark recapture program up to 1999. Biases in the mark recapture methods were identified in 1994, and methods were modified in 1995. Conclusions of a 1995 study indicated that the pooled Petersen population estimates were no longer significantly biased (Schubert and Tadey 1997; Houtman *et al.* 2000). In recent years, assessment methods have varied. Specifically, in 2000 an overflight visual survey was used to assess Sockeye escapements, in 2001 a counting tower was used, and subsequently (post-2002) an enumeration fence has been used to assess escapement. The year 2002 represents the only gap in the Birkenhead escapement time series when Sockeye escapements were not assessed. This year was gap filled using the average of the previous and subsequent generation (cycle-line average methods)(Appendices 2 & 5).

Productivity: Similar to Summer Run CUs, Lillooet-Harrison-L has exhibited systematic declines in productivity (Kalman filter Ricker *a* parameter values) since the mid-1980's (Appendix 3, Lillooet-Harrison-L, Figure 1 c). Productivity (R/S) has been particularly low recently, from the 1989 to 2005 brood years, with twelve of these years close to or below replacement (Appendix 3, Lillooet-Harrison-L, Figure 1 d). There are no freshwater or marine survival data available for this CU.

Abundance: The full stock-recruitment time series for Lillooet-Harrison-L includes the brood years 1950-2004. For Ricker model benchmark estimates (recommended model by Holt et al. 2009), a lognormally distributed prior (mean of 164,000 and sigma of 13,000), based on calculations of lake rearing capacity, was used to estimate the Ricker '*b*' parameter (Table 4; Appendix 3, Lillooet-Harrison-L, Figure 2 d). Using the standard Ricker model and the full time series (brood years 1950-2004), lower benchmarks ranged from 8,000 to 17,000 at the 10% to 90% p-levels and upper benchmarks ranged from 67,000 to 93,000 (Table 4; Appendix 3, Lillooet-Harrison-L, Figure 2 e). Statuses were assessed by comparing both the arithmetic (104,900) and the geometric (63,600) mean abundance of the last generation to the benchmarks. The resulting statuses were Green across all probability levels when comparing the arithmetic mean, and Amber across all p-levels when comparing the geometric mean (Table 4).

Given that Lillooet-Harrison-L has exhibited systematic declines in productivity, model forms that specifically consider this recent lower productivity in benchmark estimation produced higher (more biologically conservative) lower benchmarks. The recursive-Bayesian Ricker model produced the highest lower benchmarks, followed by the most truncated time series (brood years 1990-2004) Ricker model, the smoothed-Ricker model, and the less truncated (brood years 1970-2004) Ricker model. Upper benchmarks were generally lower for all models that specifically considered recent productivity, compared to the full time series standard Ricker model. Statuses for these model forms that consider recent productivity were Green for all models and probability levels when assessed using the arithmetic (104,900) mean abundance of the last generation. In contrast, statuses were Amber for the least truncated (brood years 1970-2004) Ricker model and mostly Green for all other model forms (with the exception of the 75%-90% probability levels for the recursive-Bayesian Ricker model, which were Amber) when assessed using the geometric mean (63,600) (Table 4).

Trends in Abundance: Escapements were relatively low from 1950 to 1973 (average EFS: 18,000), slightly higher from 1973 to 1985 (average EFS: 36,100), and reached a period of maximum abundance from 1986 to 2009 (average EFS: 74,400) (Appendix 3, Lillooet-Harrison-L, Figure 1 a). In many years during this most recent time period, abundances have reached as high as 200,000 EFS. Lillooet-Harrison-L has not exhibited cyclic dominance throughout the time series. Spawner success has remained high (~91%) and has not exhibited any persistent trends (Appendix 3, Lillooet-Harrison-L, Figure 1 b).

The ratio of the recent generation geometric mean to the long-term geometric mean for Lillooet-Harrison-L EFS (ratio: 1.48) is greater than the upper benchmark for this metric (ratio: 0.75) (Green status) (Table 5; Appendix 2, Lillooet-Harrison-L, Figure 2 c). In recent years (last three generations), Lillooet-Harrison-L EFS has increased (see previous paragraph). The positive slope of this recent trend (2712% change in abundance) is well above the upper benchmark for this metric (-25% change in abundance)(Green status), and there is a 2% probability that this recent trend is below the lower benchmark (Table 5; Appendix 3, Lillooet-Harrison-L, Figures 2 a & b).

Nadina-Francois-ES (merged Francois (First Run) and Francois (Second Run) CUs)

Sites: Populations that rear in Nadina Lake include Glacier Creek (located above Nadina Lake). Populations that rear in Francois Lake include Nadina River (sites include: Early and Late Nadina River), Nadina Channel, and Tagetochlain Creek (Appendix 1).

History: In recent years, Glacier Creek (upstream of Nadina Lake) was initially flown because a large Sockeye population was observed going up the falls into the lake. The system is very difficult to assess and has only been opportunistically surveyed in the last 10 to 15 years. The Glacier Creek population does not appear to be genetically distinct from the current Nadina River and channel populations. Historically, Glacier Creek Sockeye had similar timing and behaviour to the later timed (second run) components that spawn in Nadina River (see proceeding paragraph). In most years, Sockeye cannot enter Glacier Creek due to the channel diversion fence or by low water levels.

Historically, Nadina River had both earlier and later timed Sockeye populations (both were Early Summer Run timed). The first run would migrate upstream into Nadina Lake and then, to spawn, would drop back downstream (below the current channel location). A later run timing group (second run) would spawn in the current spawning channel location and was similar in behaviour and timing to the Glacier Creek population described in the previous paragraph. The Nadina Sockeye spawning channel was build in 1973 and is located south of the city of Houston, next to the Nadina River at the outlet of Nadina Lake. The channel was built to augment Nadina Sockeye and increase utilization of the Francois Lake rearing area by juveniles. After channel construction, the earlier timed (first run) Sockeye could generally no longer move up into Nadina Lake to hold prior to spawning (note: the channel diversion fence is not 100% fish tight and some fish can move upstream past it in any given year, depending on fish numbers and environmental conditions). Instead, if these fish were diverted into the channel, they generally remained in the channel due to the blockage of the bottom of the channel by a diversion weir. The first run Nadina Sockeye, however, tended to continue their behaviour of dropping back downstream to spawn, although those diverted into the channel could only drop back as far as the lower reaches of the spawning channel, rather than their original spawning locations below the spawning channel. All Sockeye (both earlier and later timed runs) not diverted into the spawning channel, now spawn below the channel. The behaviour of the first run Nadina population is relatively unique to Fraser Sockeye, as most fish that arrive first in a system generally spawn in the upper rather than lower reaches. Given the changes in behaviour and inter-spawning that now likely occurs between the first and second run Nadina River populations after channel construction, due to spatial overlap of their spawning locations, these original populations are possibly lost and replaced by a new single population (now called the Nadina-Francois-ES CU). These original populations (prior to dam construction in 1973) have been placed in the validation required category of the current CU list and titled Francois (First Run)-ES and Francois (Second Run)-ES, found in proceeding sections. Although these CUs may individually have been extirpated, there is some evidence that the original first run behaviour (arriving early and dropping back downstream to spawn), may occur in the current population structure. Research will be conducted in upcoming years to open the top of the channel during early migration to see if these Sockeye will revert to their past behaviours of migrating up to Nadina Lake.

The Nadina Channel Sockeye have experienced several years of elevated pre-spawn mortality associated with *Ichthyophthirius multifiliis,* particularly in 1978, 1987 and 1995. Although this pathogen typically does not cause disease in Sockeye Salmon, "ich" or "white spot disease" can occur if numbers of this pathogen are high due to conditions such as warm water, reduced flows and adult crowding.

Escapement Time Series: Three sites are included in the escapement time series: Early Nadina River, Late Nadina River, and Nadina Channel (Appendix 1). The time series for this CU includes only years after channel construction (post-1973), given that the population structure was different before and after this date. Despite the changes to the behaviors and spawning locations of the first and second run Nadina Sockeye after spawning channel construction, Sockeye escapements were recorded in both the early and late Nadina River sites and, therefore, both sites were included in the escapement time series. No gap filling was required for these three sites (Appendix 2). Tagetochlain Creek was excluded from the time series because it was inconsistently assessed in the 1950's & early 1960's. Glacier Creek was also excluded since it was only assessed (inconsistently) in the past 10 to 15 years.

Productivity: Similar to other Early Summer Run and Early Stuart CUs, Nadina-Francois-ES has exhibited systematic declines in productivity (Kalman filter Ricker *a* parameter values) since the mid-1960 brood years (Appendix 3, Nadina-Francois-ES, Figure 1 c). Productivity (R/S) has been particularly low recently, from the 1997 to 2005 brood years, with six of these years close to or below replacement (Appendix 3, Nadina-Francois-ES, Figure 1 d). Similar to other CUs with freshwater survival data, Nadina-Francois-ES early freshwater survival (fry/EFS) decreased consistently from the start of the time series in 1973 to the mid-1990's, and has subsequently increased (Appendix 3, Nadina-Francois-ES, Figure 1 e). Post-fry survival (recruits/fry), that includes a period of freshwater survival and marine survival, increased in the 1990's and has subsequently decreased (Appendix 3, Nadina-Francois-ES, Figure 1 f).

Abundance: The stock-recruitment time series for Nadina-Francois-ES was truncated to include only years after the construction of the spawning channel (brood years 1973-2004). This ensures consistency in the production time series and spawning area. For Ricker model benchmark estimates (recommended model by Holt et al. 2009), a uniformly distributed prior (range: 0 to 1,000,000) was used to estimate the Ricker '*b*' parameter (Table 4; Appendix 3, Nadina-Francois-ES, Figure 2 d). Using the standard Ricker model and the full time series (brood years 1973-2004), lower benchmarks ranged from 8,000 to 59,000 at the 10% to 90% probability level (p-level) and upper benchmarks ranged from 35,000 to 158,000 (Table 4; Appendix 3, Nadina-Francois-ES, Figure 2 e). Statuses were assessed by comparing both the arithmetic (9,400) and the geometric (7,000) mean abundance of the last generation to these benchmarks. The resulting statuses were Red, except at the 10% probability level, where status was Amber when assessed using the arithmetic mean (Table 4). Statuses were similar using both calculations of recent abundance, as geometric and arithmetic recent generation means were similar (only a 26% difference between arithmetic versus geometric means).

Given that Nadina-Francois-ES has exhibited systematic declines in productivity, model forms that specifically consider this recent lower productivity in benchmark estimation produced higher (more biologically conservative) lower benchmarks. The recursive-Bayesian Ricker benchmarks are not recommended for this CU, as they produce S_{max} estimates that are unrealistically high relative to other models. The smoothed-Ricker model produced the highest (most biologically conservative) lower benchmarks, followed by the most truncated (brood years 1990-2004) Ricker model. Upper benchmarks were lower for these models, compared to the full time series standard Ricker model. Statuses for these model forms that consider recent productivity were Red, with the exception of the truncated (brood years 1990-2004) Ricker model, which was Amber at the 10% p-level (Table 4).

Trends in Abundance: Nadina-Francois-ES has exhibited relatively consistent escapement throughout the time series (average EFS: 9,000), often oscillating between higher abundances on odd years (average EFS: 11,000) and lower abundances on even years (average EFS: 8,000) (Appendix 3, Nadina-Francois-ES, Figure 1 a). This CU has declined in abundance from a period of above average EFS, which occurred three generations prior to the end of the time series (22,600), to the current generation arithmetic average EFS (4,800) (Table 5; Appendix 3, Nadina-Francois-ES, Figure 1 b). Throughout the time series, spawner success has remained high (~93%) in the river and channel (90%), with the exception of 2008 when the channel had only 1% spawner success (Appendix 3, Nadina-Francois-ES, Figure 1 b).

The ratio of the recent generation geometric mean to the long-term geometric mean for Nadina-Francois-ES EFS (ratio: 0.96) is greater than the upper benchmark for this metric (ratio: 0.75) (Green status) (Table 5; Appendix 2, Nadina-Francois-ES, Figure 2 c). In recent years (last three generations), Nadina-Francois-ES EFS has decreased following a period of above average EFS (see previous paragraph). The slope of this recent trend (-37% change in abundance) is steeper than the lower benchmark for this metric (-25% change in abundance)(Red status), and there is a 69% probability that this recent trend is below the lower benchmark (Table 5; Appendix 3, Nadina-Francois-ES, Figures 2 a & b).

Nahatlatch-ES

Sites: The populations that rear in Nahatlatch Lake include Nahatlatch River and a Nahatlatch Lake spawning population; the river makes up 80% of the total on average (Appendix 1).

History: Nahatlatch-ES is relatively remote, and is located in a protected BC park. No known transplants or major human activities have occurred in this system.

Escapement Time Series: Two sites were included in the escapement time series: Nahatlatch Lake and Nahatlatch River (Appendix 1). The river assessments began in 1975 using visual surveys (peak live cumulative dead methods). Consistent lake assessments began in 1980, using standard visual survey (lake expansion) methods; there are a few years of sporadic data prior to 1980, but assessments are less reliable and did not use systematic methods. The Nahatlatch Lake estimates were gap filled for the years 1975, 1976, and 1978 using the Mean Proportion Method (gap filled values were calculated using the proportional contribution of the lake to the system using only years with assessments for both lake and river sites) (Appendices 2 & 5). In 1979 the lake estimate is included in the river abundance estimate and, therefore, gap filling for this year was not required.

Productivity: Productivity and survival could not be estimated for this CU as there are no associated recruitment data available for this CU.

Abundance: Abundance benchmarks could not be estimated for this CU as there are no associated recruitment data or spawning capacity data available for this CU.

Trends in Abundance: Abundance was lowest in Nahatlatch-ES at the start of the time series (1975-1985 average EFS: 900), highest in the middle of the time series (1986-2002 average EFS: 3,500), and has dropped again in recent years (2003-2009 average EFS: 1,100) (Appendix 3, Nahatlatch-ES, Figure 1 a). Nahatlatch-ES has not exhibited cyclic dominance within the time series. During the beginning (1975 to 1985) and end (1995 to 2009) of the time series for the Nahatlatch River site, spawner success was slightly lower and more variable (average: 94%; range: 78% to 100%) compared to the middle (1986 to 1994) component of the time series (average: 99%; range: 98% to 100%) (Appendix 3, Nahatlach-ES, Figure 1 a). The Nahatlatch Lake site shows similar trends in spawner success, but they are not used for comparison purposes due to the lower quality of data from this site.

The ratio of the recent generation geometric mean to the long-term geometric mean for Nahatlach-ES EFS (ratio: 0.55) is only slightly greater than the lower benchmark for this metric, and falls between the lower (ratio: 0.5) and upper (0.75) benchmarks (Amber status) (Table 5;

Appendix 2, Nahatlach-ES, Figure 2 c). In recent years (last three generations), Nahatlach-ES EFS has decreased following a period of average EFS (see previous paragraph) and this negative slope (-81%) is below the lower benchmark for this metric (-25% change in abundance)(Red status). There is a 100% probability that this recent trend is below the lower benchmark (Table 5; Appendix 3, Nahatlach-ES, Figures 2 a & b).

North Barriere-ES (*de novo)*

Sites: The populations that rear in North Barriere-ES include Fennell and Harper Creeks (Appendix 1). These original populations were extirpated by the construction of a downstream dam that blocked upstream adult Sockeye migration. As a result, the original population has been identified as extirpated and is documented in the proceeding CU section (North Barriere-ES (extirpated)). After the dam removal, the current CU originates from hatchery transplantation into this system. Given its hatchery origin, these populations do not represent a true CU by the WSP definition and, therefore, will be referred to as a 'CU' in quotes to distinguish it from all other Fraser Sockeye WSP CUs. Although the spawning sites in this 'CU' are relatively close in proximity to populations in the Kamloops-ES CU, they are not included in the Kamloops-ES CU since they rear in different lakes (North Barriere versus Kamloops Lake). See preceding Kamloops-ES section.

History: A dam on the Barriere River downstream of Fennell Creek obstructed Sockeye migration into this system until 1952, when it was decommissioned (Roos 1991). The original Sockeye populations that spawned upstream of the dam in Fennell and Harper Creeks were, therefore, extirpated (see proceeding section: North Barriere-ES (extirpated)). From the 1950's to 1960's, Sockeye were transplanted into the Barriere River and Fennell Creek from the Raft River (Aro 1979). Transplants to Fennell Creek were likely successful (without loss of genetic diversity), based on the genetic similarities between Fennell and its donor population (Raft) (Withler *et al.* 2000; Beacham *et al.* 2004). There is also some evidence of straying from nearby populations into Fennell Creek (Withler *et al.* 2000). Since these populations are now different from the original populations, and given that these populations rear as juveniles in North Barriere Lake, this is titled the *de novo* (new) North Barriere-ES *de novo* 'CU'.

Escapement Time Series: Only Fennell Creek was included in the escapement time series and no gap filling was required for this site (Appendices 1 & 2). Fennell Creek was consistently assessed starting in 1962 using peak live cumulative dead visual survey methods. Harper Creek was excluded, as escapements were only estimated post-1994.

Productivity: Similar to other Early Summer Run and Early Stuart CUs, North Barriere-ES (*de novo)* 'CU' has exhibited systematic declines in productivity (Kalman filter Ricker '*a*' parameter values) since the 1970 brood years (Appendix 3, North Barriere-ES (*de novo*), Figure 1 c). Productivity (R/S) has been below replacement for five years throughout the time series (Appendix 3, North Barriere-ES (*de novo)*, Figure 1 d). There are no freshwater or marine survival data available for this 'CU'.

Abundance: The stock-recruitment time series for the North Barriere-ES (*de novo*) 'CU' includes the years 1967-2004. The time series starts in 1967 to account for the removal of the Barriere dam in 1967 below Fennell Creek, to ensure consistency in the spawning area throughout the time series. For Ricker model benchmark estimates (recommended model by Holt et al. 2009), a uniformly distributed prior (range: 0 to 50,000) was used to estimate the Ricker '*b*' parameter (Table 4; Appendix 3, North Barriere-ES (*de novo)*, Figure 2 d). Using the standard Ricker model and the full time series (brood years 1967-2004), lower benchmarks ranged from 310 to

820 at the 10% to 90% p-levels, and upper benchmarks ranged from 4,100 to 6,200 (Table 4; Appendix 3, North Barriere-ES (*de novo)*, Figure 2 e). Statuses were assessed by comparing both the arithmetic (5,900) and the geometric (2,800) mean abundance of the last generation to the benchmarks. Statuses using the arithmetic mean were Green from the 10% to 75% probability levels and Amber at the 90% probability level. Statuses using the geometric mean were Amber across all probability levels (Table 4).

Given that the North Barriere-ES (*de novo)* 'CU' has exhibited systematic declines in productivity, model forms that specifically consider recent productivity in benchmark estimation produced higher (more biologically conservative) lower benchmarks. The smoothed Ricker models were not used to estimate benchmarks, as the removal of Fennell Creek from the escapement time series for the Kamlooops-ES CU occurred just prior to paper submission, and these additional analyses could not be conducted. The recursive-Bayesian model produced the highest (most biologically conservative) lower benchmarks, followed by the most truncated (brood years 1990-2004) Ricker model. Statuses using the arithmetic mean were Green at lower probability levels and Amber at higher levels. Statuses using the geometric mean were Amber across all probability levels, with the exception of the recursive Bayesian model that was Red at the 90% probability level.

Trends in Abundance: Abundances were lowest in the North Barriere-ES (*de novo)* 'CU' at the start of the time series (1967-1975 average EFS: 2,100), as the Fennell Creek Sockeye population was only beginning to build from hatchery transplants after the dam removal in 1952. This 'CU' increased in abundance to a maximum in the 1990's (1900-2000 average: 6,500), and subsequently decreased (Appendix 3, North Barriere-ES (*de novo*), Figure 1 a). North Barriere-ES (*de novo*) has not exhibited cyclic dominance throughout the time series. Spawner success was relatively high and stable throughout the time series and was, on average, 89%, with the exception of 2008 (Fennell: 20%) (Appendix 3, North Barriere-ES (*de novo*), Figure 1 a).

The ratio of the recent generation geometric mean to the long-term geometric mean for North Barriere River-ES (*de novo*) EFS (ratio: 1.58) is greater than the upper benchmark for this metric (0.75) (Green status) (Table 5; Appendix 2, North Barriere-ES (*de novo*), Figure 2 c). In recent years (last three generations), North Barriere-ES (*de novo*) EFS has decreased following a period of above average EFS (see previous paragraph), and this negative slope (-65%) is below the lower benchmark for this metric (-25% change in abundance)(Red status). There is a 100% probability that this recent trend is below the lower benchmark (Table 5; Appendix 3, North Barriere-ES (*de novo*), Figures 2 a & b).

Pitt-ES

Sites: The only site for Pitt-ES Sockeye is the Pitt River (Appendix 1).

History: The upper Pitt River is a glacially fed system originating near Isosceles Peak at an elevation of 1710 m. The river flows in a braided shifting channel across a wide, flat-bottomed valley, confined by steep mountains, and is characterized by rapids, riffles and deep pools. The river flows into Pitt Lake, which is the largest (length: 52 m) freshwater tidal lake in North America. Sockeye distribution in the upper Pitt River extends from the mouth of the river at Pitt Lake, to an area of impassable rapids 40 km upstream. Forestry is quite active in the watershed (10% of it has been logged) (Fisheries and Oceans Canada 1999).

The Pitt-ES system is extremely flashy, which can create major changes in the river channel. For example, North Boyse Creek was historically a high quality spawning location for Sockeye, until a flood event in the early 1980's changed the course of the Pitt mainstem, cutting off half of this creek from Sockeye Spawning, and flushing out most of the good spawning gravel from the remainder of this Creek (K. Peters, DFO, pers. comm.). The flashy nature of this system also creates considerable scouring action when flooding occurs. As a result, in years when high water events coincide with egg incubation, substantial egg losses can affect Sockeye production. To mitigate the effects of flooding, and associated production impacts, this CU is hatchery enhanced.

Escapement Time Series: The Pitt River site is the only site for this CU and no gap filling was required (Appendices 1 & 2). This site was assessed using mark recapture methods. The escapement time series includes Sockeye removed for hatchery enhancement.

Productivity: In contrast to other Early Summer Run and Early Stuart CUs, Pitt-ES has exhibited variable productivity (Kalman filter Ricker *a* parameter values), with high productivity between the 1960 to 1970 brood years, low productivity between the 1975 to 1990 brood years, high productivity again from the 1990 to 1995 brood years, and a subsequent decline (Appendix 3, Pitt-ES, Figure 1 c). Productivity (R/S) has been particularly low recently, from the 2000 to 2005 brood years, with productivity in all of these years falling below replacement (Appendix 3, Pitt-ES, Figure 1 d). There are no freshwater or marine survival data available for this CU.

Abundance: The full stock-recruitment time series for Pitt-ES includes the brood years 1950-2005. The Pitt escapement and recruitment time series includes fish removed for Pitt River hatchery enhancement. For Ricker model benchmark estimates (recommended model by Holt et al. 2009), a uniformly distributed prior (range: 0 to 1,500,000) was used to estimate the Ricker '*b*' parameter (Table 4; Appendix 3, Pitt-ES, Figure 2 d). Using the standard Ricker model and the full time series (brood years 1950-2004), lower benchmarks ranged from 4,000 to 9,000 at the 10% to 90% p-level, and upper benchmarks ranged from 18,000 to 26,000 (Table 4; Appendix 3, Pitt-ES, Figure 2 e). Statuses were assessed by comparing both the arithmetic (32,200) and the geometric (29,300) mean abundance of the last generation to the benchmarks. The resulting statuses were Green across all probability levels (Table 4). Statuses were similar using these two calculations of recent abundance, as geometric and arithmetic recent generation means were similar (only a 9% difference between arithmetic versus geometric means).

Given that Pitt-ES has exhibited systematic declines in productivity, model forms that specifically consider this recent lower productivity in benchmark estimation produced higher (more biologically conservative) lower benchmarks. The recursive-Bayesian Ricker model produced the highest (most biologically conservative) lower benchmarks, followed by the smoothed Ricker and truncated (brood years 1990-2004 & 1970-2004) Ricker models. Upper benchmarks for these models were similar to the standard full time series standard Ricker model, though the recursive-Bayesian Ricker benchmarks were slightly higher. Statuses for these model forms that consider recent productivity were Green for all models and probability levels, with the exception of the recursive-Bayesian Ricker model. The recursive-Bayesian Ricker model statuses shifted from Green to Amber at the 90% probability level for the arithmetic mean and the 75% p-level for the geometric mean.

Trends in Abundance: From 1950 to 1995, the Pitt-ES Sockeye escapement was relatively low (average EFS: 8,600) (Appendix 3, Pitt-ES, Figure 1 b). After 1995, escapement increased to an average of 28,000 EFS. This CU has not exhibited cyclic dominance. Spawner success has been consistently high for this CU throughout the time series (average: 96%), with the exception of 2008 (71% spawner success) (Appendix 3, Pitt-ES, Figure 1 b).

The ratio of the recent generation geometric mean to the long-term geometric mean for Pitt-ES EFS (ratio: 2.17) is over double the upper benchmark for this metric (ratio: 0.75) (Green status) (Table 5; Appendix 2, Pitt-ES, Figure 2 c). In recent years (last three generations), Pitt-ES EFS has remained relatively stable. The slope of this recent trend (0% change in abundance) is above the lower benchmark for this metric (-25% change in abundance)(Green status), and there is a 27% probability that this recent trend is below the lower benchmark (Table 5; Appendix 3, Pitt-ES, Figures 2 a & b).

Quesnel-S

Sites: (Creeks) Abbott Creek, Amos Creek, Archie Creek, Bill Miner Creek, Blue Lead Creek, Bouldery Creek, Buckingham Creek, Cameron Creek, Clearbrook Creek, Devoe Creek, East Arm - unnamed creek 1, Franks Creek, Goose Creek, Grain Creek, Hazeltine Creek, Horsefly Channel, Horsefly River, Horsefly River - Above Falls, Horsefly River – Lower, Horsefly River – Upper, Isaiah Creek, Junction Creek, Killdog Creek, Limestone Creek, Little Horsefly River, Long Creek, Lynx Creek, Marten Creek, McKinley Creek, McKinley Creek – Lower, McKinley Creek – Upper, Mitchell River, Moffat Creek, Niagara Creek, Penfold Creek, Raft Creek, Roaring River, Rock Slide, Service Creek, Spusks Creek, Sue Creek, Summit Creek, Taku Creek, Tasse Creek, Tisdall Creek, Trickle Creek, Wasko Creek, Watt Creek, Whiffle Creek, Winkley Creek. (Lake) Bear Beach – Shore, Baxter Beach, Betty Frank's – Shore, Big Slide – Shore, Big Slide, 1 km, West – shore, Bill Miner Cr. – Shore, Bill Miner Cr. - Shore 3 km west, Blue Lead Cr. – Shore, Bouldery Cr. – Shore, Bouldery Cr. - Shore 2 km east, Bowling Point, Deception Point, Devoe Creek – Shore, Double T – Shore, East Arm - Rock Slide to Peninsula Pt. Shore, East Arm - unnamed creek 2 – shore, East Arm - unnamed point, Elysia – Shore, Elysia shore - 1 km west, Franks Creek – shore, Goose Point – Shore, Goose Pt., .8 km south – shore, Goose Pt., 5 km south – shore, Grain Cr. – Shore, Horsefly Lake, Hurricane Point, Junction Shore, Killdog Creek – Shore, Lester Shore, Limestone Point – Shore, Limestone Pt, .5 km south – shore, Logger Landing, Long Cr. – Shore, Lynx Cr. – Shore, Marten Cr. – Shore, North Arm – shore (Bowling to Goose Pt.; Roaring to Deception Pt.; unnamed cove), Opa Beach, Penfold Camp Shore, Quartz Point, Quesnel Lake, Roaring Point, Roaring R. – Shore, Slate Bay, Slate Bay, 1 km east, Tasse Creek – shore, Wasko Creek – shore, Watt Cr. – Shore (Appendix 1).

History: Historically, Quesnel runs were likely in excess of 10,000,000 Sockeye on the dominant cycle years in the 1800's; escapement in 1909 was 4,000,000 (Babcock 1904). The Quesnel populations were likely the largest amongst the all Summer Run timed populations until they started to decline in the late 1800's (Roos 1991). Several key factors contributed to low abundances in the Horsefly system early in the time series (prior to 1980), including dam construction at the outlet of Quesnel Lake, placer mining, the Hells Gate landslide (1913), and droughts. Dams were constructed to hold back high water freshets for mining operations, allowing no fish to migrate past the dam into Quesnel Lake or the Horsefly River from 1898 to 1903. A fishway was in operation starting in 1905 until 1921, when the dam was removed (Roos 1991). Gold placer mining occurred in the south fork of Quesnel Lake and the Horsefly River from 1871 to 1945, and tailings from these operations were dumped into the river, covering significant areas of spawning gravel, which fish subsequently avoided during spawning. During this period of damming and mining, there was a coincidental sharp decline in the Sockeye population (Roos 1991). The 1913 Hells Gate landslide presented a further barrier to migration, particularly for later timed Quesnel Sockeye. The Quesnel Sockeye were more highly affected by the landslide than other populations because they have smaller energy reserves, and because of their spawn timing. Horsefly Sockeye spawn shortly after arriving at their spawning grounds, whereas other populations have later spawning timing (Roos 1991). Throughout the time series, droughts that de-water smaller streams, and beaver dams that present a barrier to fish migration, have both impacted the available spawning habitat in the Quesnel system. As a result, there were very few Sockeye spawners in this system from the 1930's to 1940's.

After barriers to fish migrations were eliminated, Horsefly River populations experienced strong re-building on the 1953 and 1957 cycle lines though the early to mid-1960's. Quesnel abundance started to increase notably in the 1980's, particularly on the dominant and sub-dominant cycles, reaching a peak abundance between 1992 and 2001. Increased abundance has been attributed to natural expansion and the re-invasion of remnant stocks, despite transplants (Withler *et al.* 2000) of eggs from various systems (Stellako, Bowron, Stellako, Adams, Seymour to Horsefly) to the Horsefly from the 1920's to the 1970's (Aro 1979).

A Sockeye spawning channel exists beside the Horsefly River. The channel provides an available spawning area of 15,200 m^2 and has a capacity of 12,200 females (R. Cook, DFO, pers. comm.). The initial objectives of installing the channel were to rebuild the Horsefly River Sockeye population to historic levels in the subdominant and off-years, and to supplement the dominant cycle to test Quesnel Lake's juvenile carrying capacity during "Cyclic Dominance" studies. The facility is currently operated in subdominant and off-years to rebuild the Horsefly population and increase fishing opportunities. Operation on dominant cycle years is limited (occurred on the 1989, 1993 and 2009 dominant cycle years), since returns from natural spawning areas were sufficient to test Quesnel Lake`s carrying capacity; the channel component is small relative to the natural Horsefly population.

Historically (1950's to 1970's), there has also been high pre-spawn mortality in Horsefly Sockeye, due to their earlier timing, which causes them to migrate through warmer Lower Fraser River conditions and experience higher spawning ground water temperatures. A particularly large mortality event occurred on the Horsefly in 1961, and can be attributed to a *Chondrococcus columnaris* outbreak, caused by warmer waters. In 1966, cold water was siphoned from McKinley Lake to cool McKinley Creek and control this disease, although a virulent bacterial gill disease still caused high pre-span mortality in 1969 (Roos 1991).

Escapement Time Series: The following sites were included in the escapement time series: the Cameron Creek, the Horsefly Channel, The Horsefly River, Horsefly River-Above Falls, Horsefly River-Lower, Horsefly-River-Upper, Little Horsefly River, McKinley Creek, McKinley Creek-Lower, McKinley Creek-Upper, Mitchell River and Penfold Creek sites (Appendix 1). The large number of Quesnel Lake sites (assessed using peak live cumulative dead visual surveys) were not included in the escapement time series since they represented less than 1% of the total escapement in this system post-1990 (0% pre-1990). Early in the time series, Quesnel Lake surveys were conducted and very few to no spawners were observed in the lake; abundance only starts to increase in the mid-1990's.

The Mitchell was consistently assessed throughout the time series using peak live cumulative dead visual methods, and starting in 1989 was assessed on the dominant and subdominant cycles with mark recapture methods. In 2009, the Mitchell was assessed using DIDSON methods. Two other sites included in the Mitchell time series include Cameron and Penfold Creeks, which were rolled up into the Mitchell estimate in the early time series, and broken out into their individual sites in later years. The Horsefly River, Horsefly River-Above Falls, Horsefly River-Lower, and Horsefly River-Upper were consistently assessed using peak live cumulative dead visual methods, and, in recent years (post-1980), were largely assessed with mark recapture methods. Throughout the time series, the escapement records were either rolled up into a total Horsefly River-Upper (1950-1967) or into the Horsefly River site (1993-2009), or broken down into the individual enumeration sites (1968-1992). The McKinley Creek was also consistently assessed and either rolled up into McKinley Creek (1950-1969) or broken down into the individual enumeration sites (1969-2009). In addition, there are years when McKinley estimates were rolled up into the Horsefly sites (1964, 1965 and 1981). All sites were assessed largely using peak live cumulative dead visual methods. Enumeration fences were used on the Lower McKinley and McKinley sites in recent years (sporadically post-1989). Little Horsefly River was also consistently assessed using peak live cumulative dead visual survey methods. Major gaps for all these sites occurred in 1992 (weak cycle) and 2006 (dominant cycle) for Mitchell River, and 2002 (subdominant cycle) for all other sites except Cameron Creek. Gaps were filled based on relationships between all these sites using either the two weak cycles, the dominant cycle or subdominant cycle years from 1980 to 2009, given gaps occurred during these later years when Quesnel-S abundance was significantly higher than in the early time series (Mean Proportion Method - Cyclic) (Appendices 2 & 5).

Productivity: Similar to other Summer Run CUs, the Quesnel-S has exhibited systematic declines in productivity (Kalman filter Ricker *a* parameter values) since the 1990 brood year (Appendix 3, Quesnel-S, Figure 1 c). Productivity (R/S) has been particularly low recently (1999 to 2005 brood years), with most of these years close to or below replacement (Appendix 3, Quesnel-S, Figure 1 d). Similar to other CUs with freshwater survival data, Quesnel-S early freshwater survival (fall fry/EFS) decreased from the 1970 brood years, and has subsequently increased slightly (Appendix 3, Quesnel-S, Figure 1 e). Post-fry survival (recruits/fall fry), that includes a period of freshwater survival and marine survival, increased in the 1990's and subsequently decreased (Appendix 3, Quesnel-S, Figure 1 f).

Abundance: The full stock-recruitment time series available for Quesnel-S includes the brood years 1950-2004. For Ricker model benchmark estimates (recommended model by Holt et al. 2009), a lognormally distributed prior (mean of 1,000,000 and sigma of 12,000), based on calculations of lake rearing capacity, was used to estimate the Ricker '*b*' parameter (Table 4; Appendix 3, Lillooet-Harrison-L, Figure 2 d) (Table 4; Appendix 3, Quesnel-S, Figure 2 d). Using the standard Ricker model and the full time series (brood years 1950-2004), lower benchmarks ranged from 84,000 to 168,000 at the 10% to 90% p-levels, and upper benchmarks ranged from 600,000 to 805,000 (Table 4; Appendix 3, Quesnel-S, Figure 2 e). Statuses were assessed by comparing both the arithmetic (95,800) and the geometric (51,500) mean abundance of the last generation to the benchmarks. The resulting statuses were Red across almost all probability levels, with the exception of the 10% probability level for the arithmetic mean, which was Amber (Table 4).

Given that the Quesnel-S aggregate has exhibited systematic declines in productivity, model forms that specifically consider this recent lower productivity in benchmark estimation produced higher (more biologically conservative) lower benchmarks. The recursive-Bayesian Ricker model, followed by the smoothed-Ricker and the most truncated time series (brood years 1990-2004) Ricker models produced the highest lower benchmarks. Upper benchmarks were lower for all models that specifically consider recent productivity, compared to the full time series standard Ricker model. Statuses for these model forms that consider recent productivity were Red across all probability levels, with the exception of the 10% and 25% p-levels for the less truncated (brood years 1970-2004) Ricker model evaluated with the arithmetic mean, which produced an Amber status (Table 4).

Trends in Abundance: From 1950 to 1980, the Quesnel-S escapement was relatively small (average EFS: 23,000) (Appendix 3, Quesnel-S, Figure 1 b). Escapement increased from the

1980's to 2001 (average EFS: 430,000) and was particularly high three generations prior to the end of the time series (average EFS: 585,600), and has subsequently declined to the current generation average EFS (50,700) (Table 5). Average EFS across the entire time series is 188,700 (Table 5). This CU has exhibited cyclic dominance throughout the time series, with one dominant cycle (average EFS: 500,000), one subdominant cycle that starting building in the 1980's (average EFS: 230,000) and two weak cycles (average EFS: 18,500). Spawner success has been consistently high for this CU throughout the time series, with the exception of 2008 (~60% spawner success) (Appendix 3, Quesnel-S, Figure 1 b).

The ratio of the recent generation geometric mean to the long-term geometric mean for Quesnel-S EFS (ratio: 7.7) is well above the upper benchmark for this metric (ratio: 0.75) (Green status) (Table 5; Appendix 2, Quesnel-S, Figure 2 c). In recent years (last three generations), Quesnel-S EFS has decreased following a period of above average EFS (see previous paragraph). The slope of this recent trend (-87% change in abundance) is steeper than the lower benchmark for this metric (-25% change in abundance)(Red status), and there is a 99% probability that this recent trend is below the lower benchmark (Table 5; Appendix 3, Quesnel-S, Figures 2 a & b).

Seton-L *(de novo)*

Sites: The major Late Run population that rears in Seton-L is Portage Creek (Appendix 1). Some proportion of the Early Summer timed Gates Creek and channel fish also rear in Seton Lake (Geen and Andrew 1961). Since this is a new CU originating from hatchery transplants, it technically is not a WSP CU due to its hatchery origin, therefore, 'CU' in quotes will be used to distinguish the Seton-L 'CU' from other current WSP CUs.

History: The status of the newly established population that originates from hatchery transplantation, will be evaluated in this 'CU' section (Seton-L 'CU'). Although technically the Portage Creek Late Run population would not represent a 'CU' given its hatchery origins, it is included here as it has been an established population for several decades and contributes reasonable production to the Fraser Sockeye Late Run aggregate. In 1903, the first hatchery in BC began operating on Portage Creek (Babcock 1904) near the present location of the Seton Dam. At this time, poor husbandry techniques were implicated for the declining abundance of Portage Sockeye (Geen and Andrew 1961). In 1913, the Hells Gate landslide decimated this population. In addition, water diverted from the Bridge River into Seton Lake in 1934 decreased primary productivity in this lake (Roos 1991). The original Portage population is considered extirpated (see proceeding extirpated section: Seton-S). Early observations of this original population (from old observer reports), indicate that this population was a Summer Run timed population, relative to the new hatchery origin (*de novo*) population that is Late Run timed.

In the first half of the century, various transplants were attempted in Portage Creek from multiple Fraser systems, such as Birkenhead and the Lower Adams River (Aro 1979). Genetically, the current Seton-L population is similar to the Lower Adams River, indicating that transplants from this area were most successful (Withler et al. 2000). Despite the proximity of Seton-L to Anderson-Seton-ES Sockeye during spawning, and the overlap in their rearing lakes, Seton-L is relatively genetically unique. There is also no evidence of genetic bottlenecks for Seton-L despite its genetic variability being less than the donor population (Withler et al. 2000).

A hydro facility on this system has been operational since 1956 (Roos 1991). This facility is comprised of the Seton Dam, located below the outlet of Seton Lake, and the Cayoosh Dam on Cayoosh Creek. Water is diverted by canal from Seton Lake to a powerhouse on the Fraser

River, where it is released through a tailrace located 500 m downstream of the outlet of Seton River. Since the Seton Dam presents a barrier to Sockeye migration, a fishway was constructed in concert with dam construction (Roos 1991). It has been suggested that both the tailrace and fishway may slow or impede Sockeye migration and cause physiological stress to the fish (Roscoe and Hinch 2008). Due to the downstream tailrace location, migrating adult Sockeye have been shown to stop at the outlet of the tailrace, where that are either attracted to the home-stream water or they use it as a 'cold-water' refuge. Fish may either be directly injured in the tailrace (Fretwell 1980) or indirectly suffer pre-spawn mortality due to the delay in migration from stalling at the tailrace. Success of fish departing the tailrace, entering the Seton River, and reaching the dam depends on Seton water quality, whereby higher Cayoosh Creek dilution results in higher migration failure (10-30% migration failure during IPFSC studies). Once fish enter the Seton River they must travel five kilometres upriver, ascend the Seton Dam fishway, and then migrate through Seton Lake and Anderson Lake (~50km) to the spawning grounds. One study indicated that locating the fishway entrance presents a challenge to migrating Sockeye (during experimental downstream transplants 25% of these Sockeye could not re-locate the fishway entrance) (Roscoe and Hinch 2008). Further impacts of the hydro facility include mortality (~10%) of downstream migrating smolts as they move through the dam turbines. This issue has yet to be resolved (Roos 1991).

Escapement Time Series: Only Portage Creek was included in the escapement time series and no gap filling was required (Appendices 1 & 2). This system was assessed using visual survey methods (peak live-cumulative dead) throughout the time series. Data prior to 1954 is quite sporadic, therefore, only the time series from 1965 to present (after hatchery transplants from the lower Adams River became established in the Seton system) was used in the assessment of status. This time series reflects this newly established population post-hatchery intervention.

Productivity: Similar to Early Summer Run and Early Stuart CUs, Seton-L has exhibited persistent decreases in productivity since the 1970 brood year (based on Kalman filter Ricker *a* parameter values) (Appendix 3, Seton-L, Figure 1 c). Productivity (R/S) has been particularly low recently (1999 to 2005 brood years), with three years below or close to replacement (Appendix 3, Seton-L, Figure 1 d). Freshwater and marine survival data are not available for this 'CU'.

Abundance: The full stock-recruitment time series available for Seton-L includes the brood years 1965-2004. There are considerable gaps in the early time series, and this system was only consistently assessed starting in 1965. For Ricker model benchmark estimates (recommended model by Holt et al. 2009), a uniformly distributed prior (range: 0 to 300,000) was used to estimate the Ricker '*b*' parameter (Table 4; Appendix 3, Seton-L, Figure 2 d). Using the standard Ricker model and the full time series (brood years 1967-2004), lower benchmarks ranged from 500 to 2,000 at the 10% to 90% p-levels, and upper benchmarks ranged from 6,000 to 12,000 (Table 4; Appendix 3, Seton-L, Figure 2 e). Statuses were assessed by comparing both the arithmetic (5,300) and the geometric (1,400) mean abundance of the last generation to the benchmarks. The resulting statuses were Amber across almost all probability levels, with the exception of the 90% probability level for the geometric mean, which was Red (Table 4).

Given that Seton-L has exhibited systematic declines in productivity, model forms that specifically consider this recent lower productivity in benchmark estimation produced higher (more biologically conservative) lower benchmarks, which were similar across these models. Upper benchmarks were higher for the most truncated (brood years 1990-2004) Ricker model and lower for all other models that specifically consider recent productivity, when compared to the full time series standard Ricker model. Statuses using the arithmetic (5,300) mean of recent abundances, were generally Amber, with the exception of the 90% p-level for the most truncated (brood years 1990-2004) Ricker model, which was Red in status, and the smoothed and recursive-Bayesian Ricker models, which were Green at the 10% to 25% p-levels. In contrast, statuses using the geometric (1,400) mean abundance were Amber only at the 10% to 25% p-levels and were Red at the 50% to 90% p-levels across models that consider recent productivity. The large difference between the arithmetic and geometric mean abundances (79% difference), accounts for the resulting differences in status across p-levels for these models.

Trends in Abundance: Abundance in Seton-L is relatively stable across the long-term time series (average EFS: 3,800). In recent years, however, abundance has decreased. Seton-L has exhibited cyclic dominance with one dominant cycle (average EFS: 7,900), two subdominant cycles (average EFS: 3,300), and one off cycle (average EFS: 800) (Appendix 3, Seton-L, Figure 1 a). Spawner success has remained high (~96%) and has not exhibited any persistent trends (Appendix 3, Seton-L, Figure 1 b).

The ratio of the recent generation geometric mean to the long-term geometric mean for Seton-L EFS (ratio: 0.91) is above the upper benchmark for this metric (ratio: 0.75) (Green status) (Table 5; Appendix 2, Seton-L, Figure 2 c). In recent years (last three generations), Seton-L EFS has decreased following a period of average EFS. The slope of this recent trend (-62% change in abundance) is steeper than the lower benchmark for this metric (-25% change in abundance)(Red status), and there is a 95% probability that this recent trend is below the lower benchmark (Table 5; Appendix 3, Seton-L, Figures 2 a & b).

Shuswap-ES

Site: The populations that rear in the Shuswap Lake include the Adams Channel, Adams River, Anstey River, Burton Creek, Bush Creek, Celista Creek, Craigellachie Creek, Crazy Creek, Eagle River, Hiuhill (Bear) Creek, Hunakwa Creek, Loftus Creek, McNomee Creek, Middle Shuswap River, Nikwikwaia (Gold) Creek, Onyx Creek, Pass Creek, Perry River, Ross Creek, Salmon River, Scotch Creek, Seymour River, and Yard Creek (Appendix 1). Although Upper Adams and Momich/Cayenne sites were originally included in the Shuswap-ES CU, they have now been removed. The Adams and Momich/Cayeene Creek sites have been placed in separate CUs, now considered extirpated (see proceeding sections for descriptions: Adams-ES (i.e. Upper Adams River spawners and Adams lake juvenile rearing) and Momich-ES).

History: Both Early Summer and Late Run timing populations inhabit the rearing lakes of this CU. Due to these differences in run timing and biology, spawning populations have been separated into two groups, respectively the Shuswap-ES and Shuswap Complex-L CUs (Holtby and Ciruna 2007). Within the Shuswap Lake system, no Early Summer timed Scotch Creek Sockeye historically existed on the dominant Adams River cycle (2010 cycle). In 1962, 1,023,000 eyed eggs from Seymour Creek were transplanted into Scotch Creek, producing a dominant run that coincided with the dominant Adams Late Run (2010 cycle) (Roos 1991). Anstey, Eagle and Salmon River populations were large prior to 1913, but disappeared after the Hells Gate landslide. Anstey was not enhanced by hatcheries; building of this population appears to have occurred naturally from the first Sockeye observed in this system in 1949 (Roos 1991). In that year, Sockeye were also first observed in the Eagle River (11 fish). This population was subsequently enhanced by transplants from Seymour in 1958 and 1962, which likely contributed to increased escapements by 1982. There is generally a delay in the success of transplants as they adapt to their local environment (Roos 1991). Within this system, hatchery transplants were also attempted in the Salmon, Tappen, Silver, and Silk-atwa Rivers/Creeks (1902-1931) from donor populations in Harrison, Birkenhead, Pitt, Sweltzer, and the Adams River (Aro 1979). The Salmon River population changed its dominant cycle during the 1922-42 period, and remains that way today (Roos 1991).

Escapement Time Series: Three sites were included in the escapement time series for the Shuswap Complex-ES CU: Scotch, Seymour and McNomee Creeks (Appendix 1). These sites were consistently assessed throughout the time series. Seymour and Scotch make up 75% of the total escapement in this system from 1994-2009, when other creeks/rivers were consistently assessed. Although the Eagle River and Anstey Creek were consistently assessed, they were not included in the escapement time series for reasons detailed below. All other sites had small populations, and were generally only consistently assessed post-1994; therefore a large number of these systems were excluded from escapement time series.

Seymour was the most consistently assessed site, with no gaps in the time series. Mark recapture surveys were used on large escapement years and peak live cumulative dead visual surveys were used on smaller escapement years. The Scotch time series has two missing values, prior to 1980. Until 1993, Scotch Creek was assessed with peak live cumulative dead methods (except 1990, which was a mark recapture), and as the abundance started to increase, enumeration methods switched to a fence (1994 to 2009). Gaps in the Scotch Creek escapement time series in 1951 and 1959 were filled with zeros (Appendix 2). In these years, no surveys were conducted, as the expected abundance was negligible, as seen in other off cycle years (see history of abundance in history section above). McNomee population estimates were historically rolled into Seymour, so this time series was included with no gap filling (Appendix 2).

Eagle and Anstey were excluded from the assessment of status. The survey area for Eagle was expanded in 1990 to include an area where substantial spawning occurred. As a result, Eagle escapement increased from an average of 700 total adults prior to 1990 to an average of 4,000 total adults after 1990. Due to this inconsistent methodology, its relatively small contribution to total escapement (~16%) for sites consistently assessed post-1990 (Scotch, Seymour, McNomee, Eagle, and Anstey), and its similarities to trends in the Seymour River time series, Eagle was not included in the assessment of trends in status. Anstey was also excluded because of significant gaps in the time series prior to 1990 and uncertainty in the estimates, which were due to challenges in this site's assessments. Anstey makes up, on average, only 6% of the total Scotch-Seymour-McNomee escapement, and would not have an impact on the assessment of trends if included.

Productivity: The productivity time series is relatively short for Shuswap-ES (brood years 1980-2005). Productivity (Kalman filter Ricker *a* parameter values) for this CU decreased from the 1980 to 1990 brood year, and has subsequently increased (Appendix 3, Shuswap-ES, Figure 1 c). Productivity (R/S) was particularly low from the mid-1980 to mid-1990 brood years, with four years below or close to replacement (Appendix 3, Shuswap-ES, Figure 1 d). There are no freshwater or marine survival data available for this CU.

Abundance: The full stock-recruitment time series available for Shuswap-ES includes the brood years 1980-2004. Prior to 1980, Scotch Creek was significantly enhanced on the dominant cycle of the Adams Lake run. Therefore, to ensure consistency in the time series, years prior to 1980 were not used in the stock-recruitment time series. For Ricker model benchmark estimates (recommended model by Holt et al. 2009), a uniformly distributed prior (range: 0 to 2,000,000) was used to estimate the Ricker '*b*' parameter (Table 4; Appendix 3, Shuswap-ES, Figure 2 d). Using the standard Ricker model and the full time series (brood years 1980-2004), lower benchmarks ranged from 37,000 to 253,000 at the 10% to 90% p-levels, and upper benchmarks ranged from 113,000 to 437,000 (Table 4; Appendix 3, Shuswap-ES, Figure 2 e). Statuses were assessed by comparing both the arithmetic (64,600) and the geometric (13,500) mean abundance of the last generation to the benchmarks. The resulting statuses were Red across almost all probability levels, with the exception of the 10% to 25% p-level for the arithmetic mean, which was Amber (Table 4).

Despite the fact that Shuswap-ES has exhibited systematic declines in productivity, model forms that specifically consider this recent lower productivity in benchmark estimation did not produce higher (more biologically conservative) lower benchmarks. This is possibly attributed to the shorter time series for this CU, relative to all other CUs. The recursive-Bayesian Ricker model benchmarks are not recommended for this CU as they produce S_{max} estimates that are unrealistically high relative to other models. The most truncated (brood years 1990-2004) Ricker model produced similar benchmarks to the full time series (brood years 1980-2004) standard Ricker model, and the smoothed-Ricker produced smaller lower benchmarks. Upper benchmarks follow a similar pattern to the lower benchmarks across models. Statuses for these model forms that consider recent productivity were Red across almost all models and probability levels, with the exception of the arithmetic mean at the 10% and 25% p-levels for the truncated (brood years 1990-2004) Ricker model, and the 10% to 50% p-levels for the smoothed Ricker model, which were Amber (Table 4).

Trends in Abundance: Shuswap-ES abundance was relatively small in the early time series (1950-1985 average EFS: 11,500). Abundance later increased (1986-2009: 32,600), particularly on dominant cycle years, in which abundance exceeded 100,000 EFS for three years (1990, 2002 and 2006). In particular, early in the time series (prior to the 1980's) the Seymour River dominated the trends, then in later years (post-1980's) Scotch Creek increased in abundance, equally contributing to the Shuswap-ES trend. Most Shuswap Lake rearing populations are dominant on the 2006 cycle. Seymour has consistently exhibited one dominant cycle (2006), followed by one subdominant cycle (2007) and two weak cycles (2008 & 2009). As mentioned, Scotch Creek had a different dominant cycle early in the time series (cycle 3: 2009), until hatchery transplants from Seymour River(1949-1975) (Aro 1979) built up the subdominant cycle, creating dominance in the same year as Seymour (2006). All other small creeks in the Shuswap Lake system exhibit similar cyclic dominance to Seymour.

The ratio of the recent generation geometric mean to the long-term geometric mean for Shuswap-ES EFS (ratio: 0.9) is above the upper benchmark for this metric (ratio: 0.75) (Green status) (Table 5; Appendix 2, Shuswap-ES Figure 2 c). In recent years (last three generations), Shuswap-ES EFS has decreased. The slope of this recent trend (-38% change in abundance) is steeper than the lower benchmark for this metric (-25% change in abundance)(Red status), and there is a 73% probability that this recent trend is below the lower benchmark (Table 5; Appendix 3, Shuswap-ES, Figures 2 a & b). This recent trend is attributed to an early period in the last three generations of decreased cyclicity (occurred from 1993 to 2001); this period included lower abundances in the Shuswap-ES typical dominant cycle years and higher abundances in typical weaker cycle years. This period of lower cyclic abundance produces higher geometric means relative to typical Shuswap-ES high cyclic abundance (one dominant and three weaker cycles). For highly cyclic time series, geometric means downweight the large dominant cycles relative to the three weaker cycles. Therefore, although the arithmetic averages are higher in the last generation compared to the third generation from the end (Table 5), these changes in cyclicity produced declining trends on $\log_e$ transformed four year running average abundance data.

Shuswap Complex-L

Sites: **The Shuswap Lake Complex** is comprised of five lakes: Adams Lake, Shuswap Lake, Little Shuswap Lake, Mara Lake, and Mable Lake. Populations that rear in **Adams Lake** include Adams Lake-Shore, Adams Lake-East, Adams Lake-North, Adams Lake-South, Bush Creek-Shore, Misc. East Side-Shore, Misc. North End-Shore, Misc. South End-shore, Bush Creek, Momich River, Pass Creek, Pass Creek-Shore, and Upper Adams River. **Shuswap Lake** is a large lake that can be divided into the Anstey Arm, Main Arm, Salmon Arm and Seymour Arm. Populations that rear in **Shuswap Lake-Anstey Arm** (North-East Arm) include Anstey Arm-Shore, Anstey River, Four Mile Creek-Shore, Queest Creek-shore, Vanishing Creek, Hunakwa Creek. Populations that spawn in **Shuswap Lake-Main Arm (South-West)** include Adams River, Adams River-Shore, Cruikshank Pt West-Shore, Hlina Creek-Shore, Lee Creek-Shore, Misc. North Side-Shore, Misc. South Side-Shore, Onyx Creek-Shore, Ross Creek-Shore, Scotch Creek-Shore, Adams Channel, Adams River, Hiuhill (Bear) Creek, Nikwikwaia (Gold) Creek, Onyx Creek, Ross Creek, Scotch Creek. Populations that rear in **Shuswap Lake-Salmon Arm** (South-East) include Salmon Arm-shore, Knight Creek-Shore, Misc. East Side-Shore, Misc. North Side-Shore, Misc. South Side-Shore, Reinecker Creek, Reinecker Creek Shore, Canoe Creek, Crazy Creek, Eagle River, Loftus Creek, Perry River, Salmon River, Tappen Creek and Yard Creek. Populations that rear in Shuswap Lake-Seymour Arm (North West) include miscellaneous Seymour Arm-Shore, Celista Creek, McNomee Creek, Seymour River. The only population that rears in **Little Shuswap Lake** is Little River. Populations that rear in **Mara Lake** include Mara Lake Shore, Lower Shuswap River, Cooke Creek, Kingfisher Creek and Trinity Creek. The populations that rear in **Mabel Lake** include Middle Shuswap River, Bessette Creek, Noisy Creek, Tsiuis Creek and Wap Creek (Appendix 1). In addition to these lakes, **a South Thomson River site**, located east of Kamloops Lake, was also included since it represents Shuswap Complex-L Sockeye drop outs from continued upstream migration and largely occur on dominant cycle years. A final site includes the **South Thompson River**, located east of Kamloops Lake that represents drop-outs from Shuswap Complex-L upstream migrants (Appendix 1).

History: Both Early Summer and Late Run timing populations inhabit the rearing lakes of this CU, though due to significant differences in ecology and run timing, spawning populations have been separated into two groups, respectively the Shuswap-ES and Shuswap Complex-L CUs (Holtby and Ciruna 2007). Similar to the Shuswap-ES CU, the splash dam on the Adams River and the 1913 Hells Gate landslide played a large role in the extirpation of Late Run populations that rear in Adams Lake. Current Adams Lake Late Run populations likely came from Shuswap Lake strays. The late component of the Adams Lake population is small in terms of abundance (Hume *et al.* 1996). Within Shuswap Lake, the two north arms (Seymour and Anstey) are largely undeveloped, while the two south arms (Main and Salmon) are developed for recreational and residential use. There are concerns that septic tanks in the area could leach potentially deleterious contaminants into the waterways in this southern part of the lake.

A small Sockeye population spawns in the South Thompson River, immediately upstream of Kamloops Lake. This population covaries with the Adams dominant cycle line, with negligible to no spawning on the remaining three cycles. Therefore, it is likely that this population is not unique, but rather dropouts of Adams Sockeye that were enroute to their natal Adams River. Spawners are successful at this South Thompson site and, typical of most Fraser Sockeye, emergent fry migrate to a downstream lake to rear (South Thompson fry migrate downstream to Kamloops Lake). Fry rearing in Kamloops Lake are largely from North Thompson (Raft, Fennell and North Thompson) spawning populations, with a small number of Adams Sockeye (J. Hume, DFO, pers. comm.) that likely represent the population of Adams dropouts spawning in the South Thompson. Tagging data further confirms mixing of Shuswap and South Thompson spawners on dominant cycle years.

Escapement Time Series: Twenty-eight sites were included in the escapement time series that included both rivers and creeks and lake sites. River sites included Adams River, Anstey River, Eagle River, Little River, Lower Shuswap River, Middle Shuswap River, Momich River, Pass Creek, Scotch Creek, and South Thompson River. The South Thompson River site (these Sockeye would spawn in Kamloops Lake) represents drop outs of Adams Sockeye as these fish only occur on dominant Adams cycles. Lake sites include Shuswap Lake, Adams River-Shore, Anstey Arm, Anstey River-Shore, Cruikshank Point-West-Shore, Hlina Creek-Shore, Lee Creek-Shore, Shuswap Lake-Main Arm, Shuswap Lake-Main Arm North, Shuswap Lake-Main Arm South, Onyx Creek-Shore, Ross Creek-Shore, Shuswap Lake-Salmon Arm, Shuswap Lake-Salmon Arm East, Shuswap Lake-Salmon Arm North, Shuswap Lake-Salmon Arm South, Scotch Creek-Shore, Seymour Arm. All other sites in the escapement database were excluded from the escapement time series because they were only assessed starting in the 1990's or later 2000's, and they represent negligible spawning (Appendix 1).

Adams River dominates the total abundance for this CU (70% of total EFS). The Adams River time series is complete and required no gap filling (Appendix 2). From 1950-1963, mark recapture methods were generally used to assess total abundance. From 1963-1984, the one off cycle year (cycle 3) was assessed using peak live cumulative dead visual methods (all other cycles (1,2 & 4) were assessed with mark recapture methods). From 1985 to 2009, the two off cycles (cycle 3 and cycle 4) were both assessed using peak live cumulative dead visual methods (cycle 1 and 2 were assessed using mark recapture methods). The Adams channel was excluded from the escapement time series due to sparse data (1990-2009) and negligible abundances, since this channel was designed as rearing habitat for Coho, and entry was often barricaded by beaver dams. Little River also represents a relatively high proportion of the total EFS in this CU (10% of total EFS). Little River was consistently assessed (no gaps in the time series), generally using peak live cumulative dead surveys or recovery expansions (Appendix 2). Starting in 1998, due to higher abundances (>70,000) in Little River, mark recapture methods were used for the dominant cycle, and peak live cumulative dead surveys for all other cycles. The remaining stream/river sites used to assess *trends in abundance* (Anstey, Eagle, Momich, Lower Shuswap, and Middle Shuswap Rivers, Pass and Scotch Creeks) comprised 13% of the total EFS for Shuswap Complex-L. These sites were consistently assessed on the dominant and subdominant cycles using varied assessment methods. Anstey was assessed using peak live cumulative dead visual survey methods. The number of surveys conducted in this system was generally low (1 visit per year) until 1994, when the number of visits increased (and ranged from one to six). Eagle was also generally assessed with peak live cumulative dead surveys, with the exception of a number of years (1983-1988, 1990-1992, 1994, 1998, 1999-2004 and 2006-2009) when an enumeration fence was used. Assessment methods in the Eagle River were not compromised for the Late Run populations like they were for the Early Summer, given the fish spawn in different locations. The Lower Shuswap River was also consistently assessed using peak live cumulative dead counts until the 1970's, when mark recaptures were conducted on dominant, and occasionally on subdominant, cycles. The Middle Shuswap River, Scotch Creek, Momich River and Pass Creek were all assessed using peak live cumulative dead methods. When gaps occurred in these time series a mean proportion cycle method was used to fill these (Appendices 2 & 5).

For lake sites, in early years (1950-1973) on Shuswap Lake, only Main Arm spawners were recorded. This occurred since the area between the Adams River and Little River attracts the bulk of the spawners in this CU. Therefore, crews were consistently in this area of the lake, and could easily assess shore spawners. Site resolution (number of sites recorded) increased throughout the time series for Shuswap Lake Main Arm spawners. From 1974-2001, Shuswap Lake Main Arm data were recorded as the Shuswap Lake-Main Arm site, and Shuswap Lake was no longer used as a site name. From 2002-2009 the Shuswap Lake-Main Arm site data were divided into the following nine sites: Shuswap Lake-Main Arm North, Shuswap Lake-Main Arm South, Adams River-Shore, Cruikshank Point West-Shore, Hlina Creek-Shore, Lee Creek-Shore, Onyx Creek-Shore, Ross Creek-Shore, Scotch Creek-Shore sites, and records were no longer placed in the Shuswap Lake-Main Arm site. Therefore, these nine sites, as well as the Shuswap Lake, Shuswap Lake-Main Arm and Shuswap Lake-Main Arm North and South sites were combined into the escapement time series. All Adams Lake sites had small abundances on the dominant cycle (< 4,000 total adult spawners) and had many gaps in the time series. The South Thompson River has been assessed using peak live cumulative dead visual survey methods.

Gaps in river and stream data were filled on the dominant and subdominant cycles using separate calculations for each cycle, given that individual sites varied in their proportional contribution to the total EFS depending on the cycle (dominant or subdominant). Gaps were not filled on the two weak cycles because escapement was negligible on these cycles in years when sites were assessed (frequently close to or equal to zero). No gaps were filled in the lake site data (Appendix 2). For the South Thompson River gap filling of several weak cycle years was required to complete the time series. The average of one cycle before and after each gap was used for gap filling. However, for two years with gaps, the average was calculated using data two generations removed, due to multiple consecutive gaps occurring on one cycle (Appendices 1 & 2).

Productivity: In contrast with other Early Summer Run and Early Stuart CUs, Shuswap Complex-L has not exhibited any persistent trends in productivity through time (based on Kalman filter Ricker *a* parameter values)(Appendix 3, Shuswap Complex-L, Figure 1 c). However, productivity (R/S) has been particularly low recently, from the 1998 to 2005 brood years, with five of these years below or close to replacement (Appendix 3, Harrison (U/S)-L, Figure 1 d). Shuswap Complex-L early freshwater survival (fall fry/EFS) was relatively stable throughout the time series, with the exception of high survival in the 1990's (Appendix 3, Shuswap Complex-L, Figure 1 e). Post-fry survival (recruits/fall fry), that includes a period of freshwater survival and marine survival, was relatively stable throughout the time series (Appendix 3, Shuswap Complex-L, Figure 1 f).

Abundance: The full stock-recruitment time series for Shuswap Complex-L includes the brood years 1950-2004. For Ricker model benchmark estimates (recommended model by Holt et al. 2009), a lognormally distributed prior (mean of 1,500,000 and sigma of 15,000), based on calculations of lake rearing capacity, was used to estimate the Ricker '*b*' parameter (Table 4; Appendix 3, Shuswap Complex-L, Figure 2 d). Using the standard Ricker model and the full time series (brood years 1950-2004), lower benchmarks ranged from 234,000 to 546,000 at the 10% to 90% p-level, and upper benchmarks ranged from 1,070,000 to 1,633,000 (Table 4; Appendix 3, Shuswap Complex-L, Figure 2 e). Statuses were assessed by comparing both the arithmetic (578,400) and the geometric (28,500) mean abundance of the last generation to the benchmarks. The resulting statuses were Amber across all probability levels using the arithmetic mean and Red using the geometric mean (Table 4). The large difference between the arithmetic mean and the geometric mean (100% difference) is due to the highly cyclic nature of this CU, with one dominant cycle, one subdominant cycle and two weak cycles.

In contrast to most other CUs, Shuswap Complex-L has not exhibited the systematic declines in productivity. Therefore, most model forms that specifically consider recent productivity in benchmark estimation did not produce consistently larger (more biologically conservative) benchmarks than the full time series standard Ricker model. The recursive-Bayesian Ricker model was the only model that produced higher lower benchmarks relative to the full time series standard Ricker model. All these model forms had upper benchmarks that were smaller than the full time series standard Ricker model. Statuses for these model forms that consider recent productivity were variable when assessed using the arithmetic mean abundance. Statuses were Amber for both truncated time series (brood years 1970-2004 and 1990-2004) Ricker models, Green for the smoothed Ricker model, and ranged from Green to Red (from the 10% to 90% p-levels) for the recursive-Bayesian Ricker model (Table 4). In contrast, all statuses were Red when the geometric mean abundance was compared against model benchmarks.

Trends in Abundance: Cyclic dominance is synchronous in the Shuswap Complex-L complex, and consists of a large dominant cycle (2006), followed by a much smaller subdominant cycle (2007) and two very weak cycles (2008 & 2009) (Appendix 3, Shuswap Complex-L, Figure 1 a). Abundance has been somewhat consistent in dominant cycle years for the Adams River (average: 750,000), with generally all dominant cycles above or close to 500,000 EFS. Relatively low abundances occurred in the Adams River from 1993 to 2001 (average: 140,000 EFS), peaking in 2002 (2.0 million). Exceptions to this Adams River trend include relatively high abundances post-1980 in the following Shuswap Complex-L populations: Anstey, Eagle, Pass, Middle Shuswap, Lower Shuswap and Scotch. These populations generally had two peaks in escapement, in 1990 and 2002. Similar to the Adams River, Shuswap Lake showed higher abundances starting in the 1980's, with the population declining during 1993-2001 and 2005-2009; the lake made up less than 1% of total EFS for Shuswap Complex-L on average. Momich had consistently low escapement throughout the time series (maximum: 412 EFS). Spawner success has remained high (>95%) and generally consistent, with the following exceptions: Adams River in 2000 (52%) and 2001 (90%); Momich River in 1999 (58%) and 2006 (46%), Little River in 1999 (35%), 2001 (65%) 2006 (64%) and 2007 (69%); Lower Shuswap River in 1997 (67%) and 2001 (29%), and Pass Creek in 2003 (20%) and 2006 (40%) (Appendix 3, Shuswap Complex-L, Figure 1 b).

The ratio of the recent generation geometric mean to the long-term geometric mean for Shuswap Complex-L EFS (ratio: 0.95) is greater than the upper benchmark for this metric (ratio: 0.75) (Green status) (Table 5; Appendix 2, Shuswap Complex-L, Figure 2 c). In recent years (last three generations), Shuswap Complex-L EFS has increased and the slope of this recent trend (43% change in abundance) is above the lower benchmark for this metric (-25% change in abundance)(Green status). There is only a 12% probability that this recent trend is below the lower benchmark (Table 5; Appendix 3, Shuswap Complex-L, Figures 2 a & b).

Takla-Trembleur-EStu

Sites: Two Fraser Sockeye run-timing groups (two different CUs: Early Stuart and Summer timing) rear in the Takla and Trembleur Lakes. There are 48 enumeration sites in the escapement database that have Early Stuart Run timing (Stuart-Early Stuart), including: 5 Mile Creek, 10 Mile Creek, 15 Mile Creek, 25 Mile Creek, Ankwill Creek, Baptiste Creek, Bates Creek, Bivouac Creek, Blackwater Creek, Blanchette Creek, Casamir Creek, Consolidated Creek, Crow Creek, Driftwood River, Dust Creek, Felix Creek, Fleming Creek, Forfar Creek, Forsythe Creek, French Creek, Frypan Creek, Gluske Creek, Hooker Creek, Hudson Bay Creek, Kastberg Creek, Kazchek Creek, Kotesine River, Kynoch Creek, Leo Creek, Lion Creek, McDougall Creek, Middle River (Rossette Bar), Nancut Creek, Narrows Creek, Paula Creek, Point Creek, Porter Creek, Rossette Creek, Sakeniche River, Sandpoint Creek, Shale Creek, Sinta Creek, Takla Lake-shore, Takla Lake-unnamed creek, Tarnezell Creek (same as Baptiste and Butterfield), Tildesley Creek, Unnamed Creek (placeholder for unknown names) (Appendix 1). Although Sowchea and Nahounli Creek were included in the original Stuart-EStu CU, they have now removed from the current CU list (see proceeding removed CU section: Stuart-Early Stuart), as they are considered migratory drop outs from the Takla-Trembleur-EStu CU; migratory drop-outs are not included as sites in their corresponding CUs.

History: Evidence dating as far back as 1920 indicates that the Early Stuart Run was not been large historically (Cooper and Henry 1962). Abundance was particularly low from 1962 to 1968 (average EFS: 7,000), increased to a peak of approximately 400,000 EFS in 1992, and subsequently decreased. Recent declines have occurred consistently across most streams in the CU. Studies into the decline of the Early Stuart Sockeye, conducted through the Stuart-Takla Fisheries Interaction Project, found no evidence that the spawning and incubation environment was responsible for declines in Early Stuart populations (D. Patterson, DFO, pers. comm.). Land-use changes, road densities, and stream crossings have not been proven to have negative effects on Sockeye abundance at the sub-watershed level (Macdonald *et al.* 1992). Declines have largely been attributed to the Early Stuart population's long migration route (greatest upstream migration of all Fraser Sockeye CUs), their spring (during freshet) upstream migration timing, and the increased (more extreme) water temperatures in the Fraser River post-1990. As a result, Takla-Trembleur-Early Stuart Sockeye have the highest accumulation of thermal units of any Fraser Sockeye CU, which results in fewer Takla-Trembleur-EStu Sockeye reaching the spawning grounds, due to en-route mortality, and lower spawner success (i.e. higher pre-spawn mortality) for those that survive. A decrease in marine productivity has also contributed to recent declines in the abundance of this CU.

Since the Takla-Trembleur-EStu Sockeye migrate during the spring freshet high water flows, particularly in the Fraser Canyon, they have experienced delayed migration in some years. Takla-Trembleur-EStu Sockeye were blocked downstream of Hells Gate for 15 days in 1955, due to a later than normal freshet, which resulted in very low escapement, while those that made it to the spawning grounds were in poor condition (escapement: 2,000). In 1960, this population was again 15 days late arriving on the spawning grounds, and, as a result, a large number did not reach the grounds (Holtby and Ciruna 2007). Fishways were constructed in the Fraser Canyon between 1945 and the mid-1960's, improving the ability of early timed migrants to ascend areas of difficult passage (Levy *et al.* 2007). However, further periods of low abundance occurred from 1962 to 1968, due to en-route loss, and from 1997 to 1999, due to weather conditions (Levy *et al.* 2007).

Beaver dams in the Takla-Trembleur-EStu system are an on-going problem in terms of limiting spawning habitat. Although Sockeye in this system are capable of leaping over smaller dams, larger beaver dams have presented barriers to fish passage. Most Takla-Trembleur-EStu Sockeye are thought to rear in Takla Lake, including those that spawn in the tributaries of the upper part of Middle River, near the outlet of Takla Lake (with the possible exception of Rosette) (International Pacific Salmon Fisheries Commission 1972). Small numbers of Sockeye have been observed in Sowchea Creek, but these fish are thought to be Early Stuart Sockeye drop outs from upstream migration. Sockeye numbers in Sowchea are small (average: 40 Sockeye; range: 0 to 247) and sporadic (15 years of observations only), and Sowchea does not represent a typical Early Stuart stream (Sowchea is not a small higher elevation stream but rather a larger lower elevation stream typical of Summer timed spawners not Early Stuart).

Escapement Time Series: Four key sites in the Takla/Trembleur-EStu CU have been enumerated consistently, Forfar, Gluske, Kynoch and Rossette Creeks (Appendix 1). For the first portion of the time series (1930's to late 1980's), these sites were assessed largely using peak live cumulative dead visual surveys and some mark recapture surveys, particularly in Forfar Creek (1950, 1954, 1960, 1961, 1965, 1973, 1977 and 1978); Gluske was assessed using a mark recapture in 1978, and Kynoch in 1960-1961 and 1978. Forfar, Gluske and Kynoch have been enumerated using a fence program in recent years (Gluske: 1988-2009 excluding 1993; Forfar: 1989-2009 excluding 1993 and 2007; Kynoch: 1991-2006 excluding 1993 and 1997). Data from these fenced sites, in concert with peak live cumulative dead visual surveys, have been used to develop expansion factors for all other streams assessed using peak live cumulative dead visual methods. Eight other sites that were consistently assessed using peak live cumulative dead methods include 15 Mile, 25 Mile, 5 Mile, Ankwill, Dust (mark recapture in 1981 and enumeration fences in 1997 and 2000-2006), Frypan, Shale, and Narrows Creeks. These twelve sites (Forfar, Gluske, Kynoch, Rosette, 15 Mile, 25 Mile, 5 Mile, Ankwill, Dust, Frypan, Shale, and Narrows) required negligible gap filling (Appendix 2). During the three subdominant cycles, these sites comprise, on average, 82% of the total escapement in this CU; escapement is negligible in most other sites on these cycles. On dominant years, however, these twelve sites only comprise 50% of the total escapement.

The additional sixteen sites included in the escapement time series were assessed exclusively with peak live cumulative dead methods, including: Bivouac Creek, Blackwater Creek, Consolidated Creek, Crow Creek, Driftwood River, Felix Creek, Forsythe Creek, Kastberg Creek, Kotsine River, Lion Creek, Paula Creek, Point Creek, Porter Creek, Sakeniche River, Sandpoint Creek, and Sinta Creek. These sites had numerous gaps (Appendix 2). Gaps in all streams were filled using the Mean Proportion Method - Cyclic across aggregates of sites that had correlated abundance trends (Driftwood: Blackwater, Consolidated, Driftwood, Kastberg, Kotsine, Lion, Porter, Dust, Sinta; Takla North East Arm: 5 Mile, 15 Mile, 25 Mile, Shale, Crow; Upper Trembleur: Forsythe, Ankwill, Frypan; Takla South Arm: Sandpoint, Narrows, Sakeniche, Bivouac; Trembleur:Felix, Paula, Point)(Appendices 2 & 5).

A total of twenty sites were excluded from the escapement time series. Fourteen sites were not included because they were only assessed sporadically, or they were only assessed starting in 1997 (generally). These include the following: 10 Mile Creek, Baptiste Creek, Bates Creek, Casamir Creek, Hooker Creek, Kazchek Creek, Middle River (Rossette Bar), Nancut Creek, Takla Lake-shore, Takla Lake-unnamed creek, Tarnezell Creek, Tildesly Creek, Tlitli Creek, and Unnamed Creek. An additional six sites were excluded for various reasons: Blanchette (many gaps and limited data), Fleming (methodology changed during the time series), French (many gaps and small abundances), Hudson Bay (inconsistent access), Leo (beaver dams), MacDougall (beaver dams blocked fish assess) Creeks.

Productivity: Similar to Early Summer Run CUs, Takla-Trembleur-EStu has exhibited systematic declines in productivity (Kalman filter Ricker *a* parameter values) since the mid-1960 brood years (Appendix 3, Takla-Trembleur-EStu, Figure 1 c). Productivity (R/S) has been particularly low recently, from the 1995 to 2005 brood years, with eight of these years below replacement (Appendix 3, Takla-Trembleur-EStu Figure 1 d). Early freshwater survival (fry/EFS) has been variable, increasing and decreasing throughout the time series (Appendix 3, Takla-Trembleur-EStu, Figure 1 e). Post-fry survival (recruits/fry), which includes a period of freshwater survival and marine survival, increased in the 1990's and subsequently decreased (Appendix 3, Takla-Trembleur-EStu, Figure 1 f).

Abundance: The full stock-recruitment time series available for Takla-Trembleur-EStu includes the brood years 1950-2004. For Ricker model benchmark estimates (recommended model by Holt et al. 2009), a lognormally distributed prior (mean of 600,000 and sigma of 15,000), based on calculations of lake rearing capacity, was used to estimate the Ricker '*b*' parameter (Table 4; Appendix 3, Takla-Trembleur-EStu, Figure 2 d). Using the standard Ricker model and the full time series (brood years 1950-2004), lower benchmarks ranged from 46,000 to 111,000 at the 10% to 90% p-levels, and upper benchmarks ranged from 174,000 to 302,000 (Table 4; Appendix 3, Takla-Trembleur-EStu, Figure 2 e). Statuses were assessed by comparing both the arithmetic mean (26,500) and the geometric mean (20,200) abundance of the last generation to the benchmarks. The resulting statuses were Red across all probability levels (Table 4).

Given that Takla-Trembleur-EStu has exhibited systematic declines in productivity, model forms that specifically consider this recent lower productivity in benchmark estimation produced higher (more biologically conservative) lower benchmarks. The recursive-Bayesian Ricker model, followed by the most truncated (brood years 1990-2004) Ricker model, produced the highest lower benchmarks. Upper benchmarks were similar for almost all models that specifically consider recent productivity, and were similar to the full time series standard Ricker model. One exception was the smoothed Bayesian model, which produced somewhat smaller upper benchmarks. Statuses for these model forms that consider recent productivity were Red across all models and probability levels (Table 4).

Trends in Abundance: Takla-Trembleur-EStu had relatively low escapements up to 1981 (average EFS: 30,000), increased to a peak of ~400,000 in 1992, and has subsequently declined in abundance (Appendix 3, Takla-Trembleur-EStu, Figure 1 a). A particularly low period of abundance occurred from 1962 to 1968 (average EFS: 7,000). Across the entire time series, average EFS is 40,900. This CU has further declined from a period of below average EFS, which occurred three generations prior to the end of the time series (31,000), to the current generation average EFS (13,300) (Table 5; Appendix 3, Takla-Trembleur-EStu, Figure 1 b). This CU has exhibited strong cyclic dominance throughout the time series, with the dominant cycle occurring on the 2009 cycle (one dominant cycle average EFS: 100,000 and three weaker cycles average EFS: 20,000). Spawner success has been relatively high throughout the time series (Forfar average: 90%; Gluske average: 88%; Kynoch average: 90%; Rossette average: 88%), with notably low spawner success in 1998 (range from 40-60%) and from 1978-1980 (range from 72-74%) for the four key streams in this system (Appendix 3, Takla-Trembleur-EStu, Figure 1 b).

The ratio of the recent generation geometric mean to the long-term geometric mean for Takla-Trembleur EFS (ratio: 0.58) falls between the lower (ratio: 0.5) and upper (ratio: 0.75) benchmarks for this metric (Amber status) (Table 5; Appendix 2, Takla-Trembleur-EStu, Figure 2 c). In recent years (last three generations), Takla-Trembleur-EStu EFS has decreased, following a period of below average EFS (see previous paragraph). The slope of this recent trend (-70% change in abundance) is steeper than the lower benchmark for this metric (-25% change in abundance)(Red status), and there is a 100% probability that this recent trend is below the lower benchmark (Table 5; Appendix 3, Takla-Trembleur-EStu, Figures 2 a & b).

Takla-Trembleur-Stuart-S

Sites: Two Fraser Sockeye run-timing groups (two different CUs: Summer Run and Early Stuart timing) rear in the Takla and Trembleur Lakes. Stuart Lake largely supports rearing of Summer-Run timed Sockeye with negligible to no rearing of Early Stuart Sockeye. The Summer Run timed populations that spawn at the outlet of Takla-Trembleur Lakes include Kazcheck Creek, Middle River, and Sakeniche River; populations that spawn at the outlet of Stuart Lake include Kuzkwa River, Pinchi Creek, Sowchea Creek, Tachie River, Stuart River and a Stuart Lake spawning population (Appendix 1). The Takla-Trembleur-Stuart-S CU was originally broken up into two separate CUs in previous iterations of the Fraser Sockeye CU list. However, it was confirmed that there is mixing amongst populations from these originally separate CUs and, therefore, both these CUs were combined into the current Takla-Trembleur-Stuart-S CU.

History: In the Takla-Trembleur Lakes, most Sockeye spawning occurs in the Middle River. Historically, pulpwood and sawlog harvesting, and the extension of the Pacific Great Eastern Railway caused disturbance to spawning beds in Middle River (International Pacific Salmon Fisheries Commission 1972). Currently, availability of good spawning grounds in Middle River is the main factor limiting the Takla-Trembleur-S Sockeye abundance (International Pacific Salmon Fisheries Commission 1972). Middle River was enhanced with eggs from the Birkenhead River in 1923 and Kazchek Creek received eggs from the Birkenhead, Skeena and Stuart Rivers between 1924 and 1928 (Aro 1979).

In the Stuart Lake system, the Tachie River dominates Sockeye escapements. Log driving on the Tachie River started in the 1960's (Roos 1991), similar to the Stellako River. Although this practice was discontinued on the Stellako River (Fraser-S) in 1968, it was not discontinued on the Tachie River at this time. The extent of damage to spawning grounds is unknown (Roos 1991), though it is expected to be less severe than in the Stellako River system due to differences in the physical characteristics of this system (Roos 1991). Stuart Lake has a greater capacity to rear fry than that supported by spawning sizes, indicating that the Late Stuart population may be limited in terms of available spawning grounds (Roos 1991). Hatchery transfers occurred early in this system (1907-1928), with transfers to Stuart, Pinchi, Sowchea, and Tachie from Pierre, Pinkut, Birkenhead, and the Skeena River (Aro 1979).

Escapement Time Series: For the Takla-Trembleur-Stuart-S CU, four sites were included in the escapement time series. First, in the Takla-Trembleur system, two sites were included: Middle River and Kazchek Creek (Appendix 1). Both sites have nearly complete abundance time series starting in the 1950's, and together they make up almost 100% of the total abundance in the Takla-Trembleur Lake system for years in which Sakeniche River was also assessed. Middle River was generally assessed using mark recapture methods on dominant years and peak live cumulative dead surveys on the other three cycle lines. Kazchek Creek was assessed using peak live cumulative dead methods (visual surveys). The Kazchek Creek time series was gap filled in 1984 using its relationship with Middle River, according to the Mean Proportion Method (Appendices 2 & 5). Sakeniche River was excluded from the escapement time series due to considerable enumeration gaps in the time series'; this site was also assessed using the peak live cumulative dead methodology. In years when Sakeniche was assessed, surveys were limited to one site visit only.

In addition to Takla-Trembleur Lake sites, two Stuart Lake sites were also included in the Takla-Trembleur-Stuart-S CU escapement time series: Tachie River and Kuzkwa River (Appendix 1). These two sites were consistently assessed, each has a relatively complete time series, and together they represent >96% of total escapement in years when all systems were assessed.

Tachie was consistently assessed starting in 1953. Until 1992, a mark recapture was conducted on dominant cycles, and peak live cumulative dead (air) surveys were conducted on the remaining three cycles. After 1992, mark recaptures were conducted more frequently on roughly two out of the four cycle years. Kuzkwa was assessed using peak live cumulative dead survey methods (rafting surveys or, starting in the 1960's, helicopter surveys). Kuzkwa is a larger creek that generally requires two days to assess. Usually only one survey was conducted in the early time series, coinciding with peak spawn. Up to three surveys were conducted in larger abundance years, starting in 1997. Generally, Kuzkwa has negligible abundances during the three off cycles, with larger abundances occurring only on dominant cycle years. The time series used to assess trends in abundance covers 1953 to 2009. No gap filling was required for Tachie estimates. For Kuzkwa, only 1956 was gap-filled using its proportional relationship with Tachie (Mean Proportion Method) (Appendices 2 & 5).

The remaining four sites at the outlet of Stuart Lake (Sowchea, Pinchi, Stuart River, and Stuart Lake) were excluded from the escapement time series. Sowchea Creek has not been consistently assessed because this site is not as readily accessible as other sites. Pinchi has a slightly more complete time series than Sowchea and Stuart Lake, however, most assessments were conducted inconsistently through fishery officer surveys, and data were not comparable between years until the late 1970's. Stuart Lake was assessed only once in 1958, and had a total escapement of 293; lake spawning is typically challenging to assess given that spawning can occur at depths not visible during visual surveys.

Productivity: Similar to other Summer Run CUs, the Takla-Trembleur-Stuart-S CU has exhibited systematic declines in productivity (Kalman filter Ricker *a* parameter values) since the 1990 brood year (Appendix 3, Takla-Trembleur-Stuart S, Figure 1 c). Productivity (R/S) has been particularly low recently, from the 1997 to 2005 brood years, with five of these years close to or below replacement (Appendix 3, Takla-Trembleur-Stuart S, Figure 1 d). Freshwater and marine survival data are not available for this CU.

Abundance: The full stock-recruitment time series available for Takla-Trembleur-Stuart S and includes the brood years 1950-2004. For Ricker model benchmark estimates (recommended model by Holt et al. 2009), a lognormally distributed prior (mean of 1,400,000 and sigma of 16,000), based on calculations of lake rearing capacity, was used to estimate the Ricker '*b*' parameter (Table 4; Appendix 3, Takla-Trembleur-Stuart-S, Figure 2 d). Using the standard Ricker model and the full time series (brood years 1950-2004), lower benchmarks ranged from 55,000 to 197,000 at the 10% to 90% p-levels, and upper benchmarks ranged from 343,000 to 741,000 (Table 4; Appendix 3, Takla-Trembleur-Stuart-S, Figure 2 e). Statuses were assessed by comparing both the arithmetic mean (59,100) and the geometric mean (38,400) abundance of the last generation to the benchmarks. The resulting statuses were largely Red across all probability levels, with the exception of the arithmetic mean at the 10% probability level, for which status was Amber (Table 4).

Given that Takla-Trembleur-Stuart S has exhibited systematic declines in productivity, model forms that specifically consider this recent lower productivity in benchmark estimation produced higher (more biologically conservative) lower benchmarks. The most truncated (brood years 1990-2004) Ricker model, followed by the smoothed Ricker and recursive-Bayesian Ricker model, produced the highest lower benchmarks. Upper benchmarks were similar for almost all models that specifically consider recent productivity, and were comparable to the full time series standard Ricker model. One exception was the recursive-Bayesian Ricker model, which produced much smaller upper benchmarks. Statuses for these model forms that consider recent productivity were Red across all models and probability levels (Table 4).

Trends in Abundance: Abundance in Takla-Trembleur-Stuart-S was relatively low up to 1988 (average EFS: 38,800) and was relatively high post-1988 (average EFS: 141,600). Across the entire time series, average EFS was 76,700. This CU has declined from a period of above average EFS, which occurred three generations prior to the end of the time series (122,400), to the current generation average EFS (28,100) (Table 5; Appendix 3, Takla-Trembleur-Stuart-S, Figure 1 b). The Takla-Trembleur-Stuart-S CU exhibits strong cyclic dominance, with one dominant cycle (average EFS: 250,200) and three subdominant cycles (average EFS: 19,800). Throughout the time series, spawner success has remained high (~93%) and generally consistent, with the exception of 1949-1951, which exhibited the lowest spawner success on record (average: 65%), due to high water temperatures and earlier run timing during this period.

The ratio of the recent generation geometric mean to the long-term geometric mean for Takla-Trembleur-Stuart-S EFS (ratio: 1.72) is greater than double the upper benchmark for this metric (ratio: 0.75) (Green status) (Table 5; Appendix 2, Takla-Trembleur-Stuart-S, Figure 2 c). In recent years (last three generations), Takla-Trembleur-Stuart-S EFS has decreased, following a period of above average EFS (see previous paragraph). The slope of this recent trend (-82% change in abundance) is steeper than the lower benchmark for this metric (-25% change in abundance)(Red status), and there is a 99% probability that this recent trend is below the lower benchmark (Table 5; Appendix 3, Takla-Trembleur-Stuart-S, Figures 2 a & b).

Taseko-ES

Sites: The only population to rear in Taseko Lake is the population that also spawns in Taseko Lake (Appendix 1).

History: Taseko Lake is a glacially influenced lake that has, as a result, poor fish visibility. Carcass counts are expanded based on survey effort, using methods established from studies historically conducted on Taseko Lake. Estimates are likely biased low given limitations in the number of carcasses that reach the lake surface after becoming moribund (Patterson *et al.* 2007b). Lake counts can be further compromised on survey days with heavy rain or winds that decrease the visibility of carcasses on the lake surface.

Escapement Time Series: This site has been assessed since 1949, however there are considerable gaps in the time series (Appendices 1 & 2). Gaps were filled using the cycle-line average methods (Appendices 2 & 5).

Productivity: Productivity and survival could not be estimated for this CU as there are no associated recruitment data available for this CU.

Abundance: Abundance benchmarks could not be estimated for this CU as there are no associated recruitment data or spawning capacity data available for this CU.

Trends in Abundance: The Taseko Lake Sockeye population is small in abundance (average EFS: 1,300) (Appendix 3, Taseko-ES, Figure 1 b). This population has decreased in abundance from a peak period of 2,900 EFS (1950-1964) to an average of 376 EFS (1990-2009). This CU has not exhibited cyclic dominance, and throughout the time series spawner success has remained high (~93%)(Appendix 3, Taseko-ES, Figure 1 b).

For all calculations, the time series of this CU was limited to include only surveyed years. There are considerable gaps in the middle of the time series that cannot be gap filled; therefore, only the early time series (1950-1968) and the recent period (1993-2009) were used. The ratio of the recent generation geometric mean to the long-term geometric mean for Taseko-ES EFS (ratio: 0.32) falls below the lower benchmark (ratio: 0.5) for this metric (Red status) (Table 5; Appendix 2, Taseko-ES, Figure 2 c). In recent years (last three generations), Taseko-ES has further decreased following a period of below average EFS (see previous paragraph). The slope of this recent trend (-76% change in abundance) is steeper than the lower benchmark for this metric (-25% change in abundance)(Red status), and there is a 97% probability that this recent trend is below the lower benchmark (Table 5; Appendix 3, Taseko-ES, Figures 2 a & b). The average size of this CU is small (average ETS: 2,300).

Widgeon (River-Type)

Sites: The Widgeon CU is a river-type population and includes only one population: Widgeon Creek (Appendix 1).

History: Widgeon (River-Type) Sockeye are possibly the most unique CU in the Fraser Watershed. This population is adapted to the tidal conditions of Widgeon Slough. The fish move back and forth between Pitt Lake and Widgeon Slough with the tides, moving into the slough to spawn on high tides and moving into Pitt Lake on low tides. Due to consistent Sockeye movement into the slough, a channel has developed through which they migrate, facilitating the counting of fish. Sockeye also move into areas in Widgeon Slough where eel grass covers the spawning gravel, though it is unclear whether they do this for protection from predators (defense) or for spawning. Water levels are very low during low tide (de-watered) with only sufficient cover for egg incubation, therefore, atypical of the Sockeye species, females cannot remain with their nests until they die. Overall, the spawning area is very small (~100 m in length) and visibility of Sockeye is good. Widgeon Sockeye are similar to Harrison (River-Type) Sockeye in that they migrate to the ocean after gravel emergence and do not rear in lakes as juveniles. Widgeon (River-Type) Sockeye are also the smallest adults in the watershed.

Escapement Time Series: Widgeon Slough has been assessed consistently using peak live cumulative dead visual (foot) surveys. There are three gaps in the time series where incomplete surveys were conducted (Appendices 1 & 2). These gaps were filled using the cycle-line average (Appendices2 & 5).

Productivity: Productivity and survival could not be estimated for this CU as there are no associated recruitment data available for this CU.

Abundance: Abundance benchmarks could not be estimated for this CU as there are no associated recruitment data or spawning capacity data available for this CU.

Trends in Abundance: Widgeon has an extremely small population (average EFS: 300). This population has decreased in abundance from a peak period of 400 EFS (1950-1989) to a more recent average of 120 EFS (1990-2009). In 2009, the abundance increased to 800 EFS (Appendix 3, Widgeon (River-Type), Figure 1 b). Throughout the time series spawner success has remained high (~96%).

The ratio of the recent generation geometric mean to the long-term geometric mean for Widgeon (River-Type) EFS (ratio: 0.35) falls below the lower benchmark (ratio: 0.5) for this metric (Red status) (Table 5; Appendix 2, Widgeon (River-Type), Figure 2 c). In recent years (last three generations), Widgeon (River-Type) has increased following a period of below average abundance, including a large increase in 2009 (see previous paragraph). The slope of this recent trend (437% change in abundance) is above the upper benchmark for this metric (-15% change in abundance)(Green status), and there is a 0% probability that this recent trend is below this lower benchmark (Table 5; Appendix 3, Widgeon (River-Type), Figures 2 a & b). The average size of this CU is small (average ETS: 625).

Six Validation Required Conservation Units

Cariboo-S (extirpated?)

There are four recent years of observations (post-2002) of Sockeye in the Upper Cariboo River above Cariboo Lake, obtained from escapement enumeration programs for Chinook and Coho (e.g. 300 Sockeye were observed at this site in 2007). At present, there is not sufficient evidence to confirm whether or not this is a current CU. Future assessments and biological data are required.

Francois (First Run)-ES and Francois (Second Run)-ES (extirpated?)

Historically, Nadina River had both an earlier and later timed Sockeye run (both were early summer timing). The first run would migrate upstream into Nadina Lake and then, to spawn, would drop back downstream (below the current channel location). A later run timing group (second run) would spawn in the current spawning channel location. A Glacier Creek (upstream of Nadina Lake) population is included in the Francois (First Run)-ES CU, since both populations had similar timing and behaviour. The Nadina Sockeye spawning channel was build in 1973 and is located south of the city of Houston, next to the Nadina River at the outlet of Nadina Lake. The channel was built to augment Nadina Sockeye and increase utilization of the Francois Lake rearing area by juveniles. After channel construction, the earlier timed (first run) Nadina River spawning Sockeye could generally no longer move upstream into Nadina Lake to hold prior to spawning (note: the channel diversion fence is not 100% fish tight and some fish can move upstream past it in any given year, depending on fish numbers and environmental conditions). Instead, if these fish were diverted into the channel, they generally remained in the channel due to the blockage of the bottom of the channel by a diversion weir. The first run Sockeye, however, tended to continue their behaviour of dropping back downstream to spawn, although for those diverted into the channel they could only drop back as far as the lower reaches of the spawning channel rather than their original spawning locations below the spawning channel. All Sockeye (both earlier and later timed runs) not diverted into the spawning channel now spawn below the channel. This behaviour of the first run population is relatively unique to Fraser Sockeye, as most fish that arrive first in a system generally spawn in the upper rather than lower reaches. The original first run behaviour (arriving early and dropping back downstream to spawn), appears to somewhat have been retained in the current population structure. Research will be conducted in upcoming years to open the top of the channel during early migration to see if these Sockeye will revert to their past beavers of migrating up to Nadina Lake prior to spawning. Given that these two pre-channel populations were distinct in behaviour and spawning locations, they are now identified in the current CU list as two separate CUs: Francois (First Run)-ES and Francois (Second Run)-ES, in addition to the post-channel Nadina-Francois-ES CU that is a mixture of these two runs.

Indian/Kruger

The last Sockeye observations in this system were in 1986, from a weir-based escapement enumeration program. It is probable that there were Sockeye in the Indian/Kruger Lakes historically but it is unknown if they are currently present. There are a few reported Sockeye in

UBC records, however these have not been verified as anadromous Sockeye and may, in fact, be kokanee. There are several observations of Chinook in this system, indicating that the lakes are accessible to anadromous salmon. Since assessments have been extremely infrequent, further assessments (DFO assessments are planned here for 2011) are required to confirm the persistence of this population.

Mid-Fraser River (River-Type)

This CU includes the following sites: Bridge River, Nechako River and Yalakom River. Other sites, West Road (Blackwater) and Cariboo Rivers, require additional verification to confirm they are persistent river-type populations. There are a number of other sites that were originally included in this CU that are now confirmed to be upstream migration drop-outs and, therefore, not persistent river-type populations (e.g. Hawks Creek). Continued DNA analysis, scale life-history analysis (to confirm river-type life-history), and escapement enumeration is required at all identified sites to assess whether or not this is a current CU.

Upper Fraser (River-Type)

This CU includes the following sites: Bridge River, Indian Point Creek (three observations from a Weir and one in 2010 and 1985 and 1986 from CN fence and 1951), Lower Bowron River, Fraser River above Tete Jaune Creek, Holmes and McGregor Rivers, and Swift and Slim Creeks. There has only been one observation of Sockeye in Tete Jaune Creek in the Upper Fraser, and this was observed opportunistically during a Chinook survey. Chinook are consistently assessed in this system, therefore, indicating that Sockeye have not likely been present in other years. This CU is a placeholder until more data can be collected to confirm whether these are persistent river-type populations.

Eight Extirpated Conservation Units

Adams-ES (i.e. Upper Adams River)

Originally, the Upper Adams River spawning population was placed in the Shuswap-ES CU (see preceding section: Shuswap-ES). However given the separate rearing lake (Adams Lake) and distance from Shuswap Lake, this population was moved to this separate CU (Adams-ES), similar to the proceeding Momich-ES CU. The Adams-ES CU includes a single population of Early Summer timed Sockeye that spawn in the Upper Adams River and rear in Adams Lake as juveniles. The original population was thought to have been extirpated by the combined effects of the Fraser Canyon's Hells Gate landslide in 1913, and splash damming on the lower Adams River (1908-1940), which severely obstructed Sockeye access through the Fraser Canyon and into the Adams Lake. Although hatchery enhancement of the Upper Adams River from 1948-1980, using largely Seymour River (and to a lesser extent Taseko and Cayenne) Sockeye (Roos 1991; Withler *et al.* 2000) occurred, enhancement has been largely unsuccessful as these populations have remained extremely small (for many off cycle years since 1950, no Sockeye have been observed). Currently, Sockeye in the Upper Adams River population are highly genetically related to the donor (Seymour River) population, although some genetic differences exist. The original Adams-ES CU is considered extirpated and has been replaced by a small population that originated and has been maintained by hatchery enhancement. It is unclear if this new hatchery-origin population will be self-sustaining and, therefore, it is not at this time considered a *de novo* 'CU'.

Alouette-ES

An Early Summer timed run (April-July migration) of anadromous Sockeye salmon spawned (September-November) in the mainstem of the Alouette River and reared in Alouette Lake prior to the construction of the hydroelectric dam (1925-1928) on this system (Gaboury and Bocking 2004). After its construction, the dam blocked fish passage and eliminated this run of Sockeye salmon; at this time Alouette-ES was considered extirpated and unrecoverable. However, a spillway was constructed in 1985, and, as a result of recent experimentation in flow regimes (2005-2009) over the dam (spillway releases), some Sockeye smolts (from reservoir kokanee) emigrated from the Alouette reservoir. These Sockeye returned to the dam at the outlet of the Alouette reservoir years later as adults (confirmed to have originated from emigrating Alouette smolts), after a period of ocean residence (Mathews and Bocking 2007; Godbout *et al.* 2011). Therefore, it appears that the resident kokanee that originated from the pre-dam anadromous Sockeye have retained their anadromous life-history. Recovery of Alouette-ES Sockeye requires the continuation of spill regimes that permit outmigration of Sockeye smolts (currently occurs each spring as part of the Alouette water use plan) and the manual trucking (Trap & Truck Program) of returning adult fish back into the reservoir (Balcke 2009), or, alternatively, the construction of a fishway for adult migration (Gaboury and Bocking 2004). The Alouette-ES CU is currently not a self-sustaining anadromous Sockeye Run, and therefore, is technically considered an extirpated CU despite the occurrence of resident kokanee that have retained the ability to re-anadromize. The restoration of anadromous fish runs, where practical, is a key objective of the Bridge-Coastal Fish and Wildlife Restoration Program (BCRP).

Coquitlam-ES

An Early Summer timed run of anadromous Sockeye salmon reared in Coquitlam Lake prior to the construction of a hydroelectric dam (1914) on this system. The Coquitlam Reservoir is now one of three lakes that contributes to the Vancouver Water District municipal water supply (Fisheries and Oceans Canada 1999). After its construction, the dam blocked fish passage and, as a result, eliminated this run of anadromous Sockeye salmon; at this time the population was considered extirpated and unrecoverable. However, in recent years (2005-2009) due to some experimentation in flow regimes over the dam (spillway releases), some Sockeye smolts (from reservoir kokanee) emigrated from the Coquitlam reservoir. These fish returned to the dam at the outlet of the Coquitlam reservoir years later as adults, after a period of ocean residence (Godbout *et al.* 2011). Both genetic and gill raker analyses of kokanee and volitional (fish spilled over the dam) Sockeye smolts in the Coquitlam reservoir indicate that these fish are similar, and that the kokanee have been recently derived from anadromous Sockeye. Genetic and freshwater isotopic signature analysis confirms that anadromous sockeye returns to the Coquitlam River in recent years originate from Coquitlam reservoir kokanee populations (Godbout *et al.* 2011). Coquitlam Sockeye are closely related to nearby Pitt River Sockeye, suggesting a common colonizing population, and straying between these populations prior to dam construction (Nelson and Wood, 2007). Recovery of Coquitlam-ES Sockeye would require spill regimes that would permit outmigration of Sockeye smolts, and the manual trucking of returning adult fish back into the reservoir on the other side of the dam. The Coquitlam-ES CU is currently not a self-sustaining anadromous Sockeye Run, and therefore, is technically considered an extirpated CU despite the occurrence of resident kokanee that have retained the ability to re-anadromize. The restoration of anadromous fish runs, where practical, is a key objective of the Bridge-Coastal Fish and Wildlife Restoration Program (BCRP).

Fraser-ES

This CU includes two sites: Endako River and Ormonde Creek. These populations are likely extirpated and were never large since the substrate is of poor quality for salmon and there is much better gravel for Sockeye spawning in other locations. Sockeye are no longer present in the Endako River and the early summer timed component of Ormonde Creek has not been observed since the 1970's. Note there is a later timed (Summer Run timing) component of Sockeye that also spawns in Ormonde Creek that is part of the preceding Francois-Fraser-S section.

Kawkawa-L

Kawkawa Lake was dammed in the past (date unknown) and, as a result, has not been accessible to spawning Sockeye since its construction. There may have been anadromous Sockeye in this system prior to damming, although this has not been confirmed. Roos (1991) reported that Sockeye were observed in Kawkawa Lake during the years of the Hells Gate fishway construction (late 1920's and early 1930's) after the landslide (1913). However, these spawning fish, were reported to be Adams River Sockeye that had dropped out of upstream migration before Hells Gate. Currently, kokanee (non-anadromous Sockeye) do occupy the lake. There have been no experiments, similar to the extirpated Alouette-ES and Coquitlam-ES CU, to explore whether the anadromous life-history occurs in the resident kokanee population. Although this CU has been included in the current CU list and classified as extirpated, there is only limited evidence currently to suggest that this was a persistent lake-type population in the Fraser watershed.

Momich-ES

Similar to the Adams-ES extirpated CU, Momich and Cayenne Creek populations were previously placed in the Shuswap-ES CU (see preceding section: Shuswap-ES). However given the separate rearing lake (Momich Lake) and distance from Shuswap Lake, these populations were moved to a separate CU (Momich-ES), similar to the Adams-ES (i.e. Upper Adams River) CU. Also like the Adams-ES CU, the Early Summer timed populations that spawn in Momich River and Cayenne Creek and likely rear in Momich Lake were thought to have been extirpated by the combined effects of the Fraser Canyon's Hells Gate landslide in 1913 and splash damming on the lower Adams River (1908-1940), which severely obstructed Sockeye access through the Fraser Canyon and into Adams and Momich Lakes. The Momich River Sockeye population re-appeared in 1960, although the origin of this population is not well known. There were no hatchery transplants directly into this system and it is unlikely Momich River Sockeye are from their original population, since none should have survived the earlier obstructions (Hells Gate landslide and the Adams splash dam). It is more likely that the current Momich Sockeye population originated as strays from egg transplants from the Seymour River to the Upper Adams River (Roos 1991). Similarly, the adjacent Cayenne Creek population was also first observed again in 1960, and also likely originated from earlier transplants of Seymour and/or Taseko eggs and juveniles into Adams Lake (cited from Williams 1987) (Withler *et al.* 2000). Adams and Momich Lake populations combined are now genetically distinct, as a result of genetic drift or founder effects (Withler *et al.* 2000). In response to low returns of Adams and Momich Lake populations in 1992, restoration efforts have enhanced the offspring of this cycle year, through a combination of reduced fishing, hatchery releases and nutrient enrichment of the lake nursery area (Hume *et al.* 2003). The original Momich-ES CU is considered extirpated and replaced by a small population that originates from, and has been maintained by, hatchery enhancement. Since it is unclear if this new population will be self-sustaining, there is currently no new *de novo* 'CU' identified.

North Barriere-ES

Since the lower Barriere River, downstream of Fennell Creek, was dammed up to 1952 (after which it was decommissioned), Sockeye populations that spawned upstream of the dam (Harper, Fennell Creeks and Upper Barriere River) became extirpated. Since juveniles from these populations would have reared in the North Barriere Lake, these populations were not included in the nearby Kamloops-ES CU, but instead placed in the North Barriere-ES CU. There is an identically named North Barriere-ES (*de novo*) 'CU' that refers to the re-established Sockeye populations in these sites due to hatchery enhancement. These populations are different from the original extirpated populations.

Seton-S

This was a population of Summer Run timed Sockeye that spawned in Portage Creek and reared as juveniles in Seton Lake. A number of factors contributed to the extirpation of this CU including early hatchery programs, the 1913 Hells Gate landslide, and water diversion. Specifically, in 1903 the first hatchery in BC began operating on Portage Creek (Babcock 1904) near the present location of the Seton Dam. At this time, poor husbandry techniques were implicated for the declining abundance of Portage Sockeye (Geen and Andrew 1961). In 1913, the Hells Gate landslide decimated what remained of this population. In addition, water diverted from the Bridge River into Seton Lake in 1934 decreased primary productivity in this lake (Roos 1991). As a result, the original Portage Summer Run timed population is considered extirpated. Due to hatchery transplantation into the Portage Creek from multiple Fraser systems, such as Birkenhead and the Lower Adams River (Aro 1979), a new population was established and is described and assessed in the preceding section: Seton-L (*de novo*). Technically this new population is not a CU since it originates from hatchery intervention, however, it is included as a 'CU' in biological status assessments given it contributes reasonable production to the Late Run Fraser Sockeye aggregate.

Removed (6) Conservation Units

There are a number CUs that occurred in previous versions of the Fraser Sockeye CU list that after subsequent review of the data and associated information, it was decided that they were not, in fact, valid CUs. As a result, these CUs were removed from the current Fraser Sockeye CU list. Rationale for their removal from the CU list are described below.

Boundary Bay (River-Type)

Although this is not a Fraser Sockeye CU, it was included here since it occurs near the Fraser confluence with the Strait of Georgia. There is only one recent observation of Sockeye for this CU. Currently, this observation (recorded in the escapement database) has not been verified or confirmed in the historical records. Further, anecdotal information suggests that when Sockeye are observed in this system they are associated with the dominant Adams run (Shuswap Complex-L CU), and, therefore, are likely strays from the Fraser system. Therefore, this CU was removed from the Fraser Sockeye CU list.

Carpenter Lake

Prior to the construction of the Terzaghi Dam on the Bridge River, Sockeye were enumerated at several sites upstream of the dam. In the original CU description, it was presumed that there was an antecedent lake to Carpenter Lake and that those Sockeye were lake-type. However, after further investigations no evidence exists that confirms that there was a lake utilized by sockeye in the Bridge system (B. Holtby, DFO, pers. comm.). Consequently, the Carpenter Lake CU has been removed from the Fraser Sockeye CU list.

Hayward Lake

There were several errors in characterizing this lake system as a Sockeye lake-type CU. First, Hayward Lake was formed by the construction of the Ruskin Dam in the Stave River Canyon in 1930. Therefore, the Hayward Lake name is erroneous as it is, in fact, named the Ruskin or Hayward Reservoir. Natural falls occur in this system and are likely impassable to all fish, therefore, no anadromous Sockeye would have spawned in this system. Upstream of the Ruskin/Hayward Reservoir, in Stave Lake, sediment cores and isotopic analyses also found no evidence of anadromous salmonids in this system (B. Holtby, DFO, pers. comm.). Given that historically there was no Ruskin/Hayward reservoir, there would be no Hayward Lake CU for Fraser Sockeye. Further, since there is no evidence of anadromous Sockeye in Stave Lake, there also would be no Stave Lake CU. Therefore, this CU was removed from the Fraser Sockeye CU list.

Fraser Canyon (River-Type)

All observations of Sockeye at sites in this CU (e.g. American, Emory, Silverhope, Spuzzum and Yale Creeks and the Coquihalla River) largely occurred in the late-1930's during the fishway construction at Hells Gate after the Hells Gate landslide (1913) that blocked fish passage. Hells Gate represents a location of migratory challenge for Sockeye upstream migration, given high discharges and flows that occur at this constricted location in the Fraser, particularly during the spring freshets. Given the timing of these early Sockeye observations in Fraser Canyon streams and rivers, these fish are likely migratory drop-outs from upstream lake-type CUs and do not represent components of a unique and distinct river-type CU. Outside of these years, there is limited data for these populations, as these sites were only assessed during Pink years or years in which there were reported difficult migration conditions coinciding with the dominant cycle Adams River Sockeye run. Almost all field notes for these creeks and rivers indicate that these Sockeye observations were likely attributed to lake-type CU drop outs due to upstream migratory challenges or were associated with the large dominant Adams run. Given this information, it was concluded that these Sockeye observations in Fraser Canyon streams do not represent a unique Fraser Sockeye river-type populations in the Fraser watershed and, therefore, this CU was removed from the Fraser Sockeye CU list.

Thompson (River-Type)

The sites in this CU include the mainstem of the Thompson River and Deadman Creek. The Thompson River was only assessed in Pink (odd) years and Deadman Creek was assessed once by a fence operated by the Skeetchestn Indian Band. The Sockeye observations in these sites have been confirmed drop outs from the Kamloops-ES and Shuswap-ES CU. Therefore, this CU was removed from the Fraser Sockeye CU list.

Stuart-Early Stuart

This was a CU in earlier versions of the CU list. However, this CU was dropped from current iterations as sites associated with this CU are drop outs from upstream migrating Takla-Trembleur-EStu Sockeye (see proceeding section: Takla-Trembleur-EStu). There were two sites associated with the Stuart-EStu CU, both of which have only one year of data (Nahounli Creek) or negligible escapement data (Sowchea Creek). The population in Nahounli Creek is not persistent, and was only surveyed in 1951. There are sixteen escapement records for Sowchea Creek, occurring in 1941, 1951, 1955, 1956, 1960, 1970, 1974, 2001, and during 2003-2009. Sockeye are observed in these creeks only when spawner abundance in the Takla-Trembleur CU is high or migration conditions have been stressful (e.g. warmer water conditions). These populations are not genetically distinct from the Takla-Trembleur-EStu CU and are not persistent. Therefore, this CU was removed from the Fraser Sockeye CU list.

CONCLUSIONS

For Fraser Sockeye CUs, *abundance* benchmarks (unique to each CU) were estimated across a range of model forms and probability levels and *trends in abundance* benchmarks (common across each CU) were modified from those recommended by Holt et al. (2009). For each benchmark combination, statuses were then assessed for each CU. In addition, *abundance* metric statuses were also assessed by comparing two estimates of recent abundances (arithmetic versus geometric means) against the range of lower and upper benchmark combinations for each CU. Given the number of metrics and the evaluation of uncertainty in *abundance* benchmarks, statuses for a CU could be comprised of all three WSP status zones (Red, Amber and Green) (Tables 4 & 5).

For abundance metrics, both the structural (different Ricker model forms) and stochastic (probability distributions) uncertainty in benchmarks are presented in Table 4. Lower and upper Fraser Sockeye *abundance* benchmarks were estimated first using the full stock-recruitment time series standard Ricker model (that assumes constant productivity) within a Bayesian framework. Since most Fraser Sockeye CUs have exhibited declines in productivity in recent decades, Ricker model forms that assume non-stationary productivity (truncated stock-recruitment time series Ricker models, smoothed-Ricker models and recursive-Bayesian Ricker models) were also used to estimate *abundance* benchmarks. For CUs that have exhibited recent productivity declines, model forms that emphasize this recent productivity in benchmark estimation, generally produced larger (more biologically conservative) lower benchmarks, compared to the standard (full time-series) Ricker model. This result is attributed to the negative covariation between the WSP *abundance* metric lower benchmark (S_{gen}: recovery to S_{msy} in one generation under equilibrium conditions) and the Ricker model's intrinsic productivity ('*a*') parameter (Holt and Bradford 2011). Recent simulation results reported that while S_{gen} benchmarks increase as a population's intrinsic productivity decreases from moderate to low, other benchmarks (e.g. benchmarks that are percentages of S_{msy}) that do not covary with productivity, don't change significantly (Holt and Bradford 2011). Therefore, during periods of reduced productivity, larger lower benchmarks (estimated from Ricker model forms that emphasize recent lower productivity) may assist with protecting CUs from extirpation, depending on how often benchmarks are re-estimated and how these results are applied to harvest management.

In contrast to *abundance* lower (S_{gen}) benchmarks, upper benchmarks (80% S_{msy}) do not covary with productivity (Holt and Bradford 2011). Therefore, although lower benchmarks generally increase when using Ricker model forms that emphasize recent lower productivity, upper benchmarks are generally similar (or in a number of cases even smaller) to those estimated with the full time-series Ricker model. Therefore, changes in CU statuses across model forms are largely driven by changes in the lower benchmarks or differences in the method used (arithmetic versus geometric mean) to calculate the average abundance in the last generation, rather than changes in the upper benchmark.

For all models, prior information on the carrying capacity parameter ('*b*' parameter) was used, if available and appropriate for a CU. The current paper updates the photosynthetic rate (PR) model's estimates of spawners at maximum juvenile production (S_{max})(Hume et al. 1996; Shortreed et al. 2000; Cox-Rogers et al. 2010) by considering competitors to Fraser Sockeye juveniles in rearing lakes. The current paper uses these results (pairing appropriate lake estimates with corresponding CUs) to estimate the mean of the Ricker model carrying capacity (Ricker '*b*') parameter for use as a Bayesian prior distribution (Table 3). The standard deviation (sigma) of this prior distribution, however, was selected to exceed a CU's range of spawners observed in the escapement time series. For future analyses, the current paper's S_{max} estimates could be updated using Bodtker et al.'s (2007) Bayesian PR method that accounts for uncertainties in the PR model. The resultant S_{max} probability distributions could then be used as carrying capacity (Ricker '*b*') parameter prior distributions instead of our current approach to estimate priors.

Prior to inclusion of PR model results as carrying capacity prior distributions, freshwater capacity estimates based on stock-recruitment data and PR model results were compared for each CU. Where informative PR model carrying capacity prior information was used, Bayesian posterior distributions did not change considerably when compared to uninformative distributions. For some CUs, such as Shuswap-ES, PR model spawner abundances at maximum juvenile production (S_{max}) were much greater than those estimated from stock-recruitment data alone, indicating that other mechanisms besides lake-rearing habitat were limiting juvenile production. In addition juvenile production for river-type CUs that do not rear in lakes but migrate to the ocean after gravel emergence would also not be limited by lake-rearing habitats. These considerations were evaluated prior to the inclusion of PR S_{max} estimates as Ricker model carrying capacity priors (Tables 3 & 4). Bodtker et al. (2007) indicated that such considerations are important when combining information from different sources (e.g. stock-recruitment data with PR data). Specifically, if results from these different sources are contradictory and informative, the combined capacity estimate may be incorrect since they may not be supported by either the stock-recruitment data or the PR data (Bodtker 2007). To improve benchmark estimation for CUs where spawning habitat may be driving freshwater production, rather than lake rearing habitat, spawning habitat capacity estimates should also be developed for Fraser Sockeye CUs.

For most CUs, the method used to average last generation abundances (geometric versus arithmetic means) generally did not affect *abundance* statuses, except for CUs that have exhibited highly variable abundances across the four cycle lines in the last generation (e.g. Shuswap-ES, Chilliwack-ES, Shuswap Complex-L, and Seton-L (*de novo*)). For these exceptions, the geometric mean produced much lower recent abundance estimates, since the high abundance (dominant) cycle line is down weighted considerably in this calculation, relative to the three weak cycle years. Since abundances for highly cyclic CUs are lognormally distributed (with generally three relatively low abundance cycle years and one significantly larger abundance cycle year, per generation), the geometric mean, theoretically, should more appropriately reflect the central tendency of a CU's abundance. However, for Fraser Sockeye, due to the two-tiered escapement enumeration program that uses higher precision assessment methods on escapements greater than 75,000 (increased from 25,000 in 2004), the higher abundance years generally are estimated more precisely than the lower abundance years, depending on the CU and low abundance population sizes. Specifically, visually assessed escapements tend to be biased low for populations predicted to be close to this 75,000 value, based on recent calibration work (K. Benner, DFO, pers. comm.). Therefore, geometric means of recent abundances may underestimate average abundance due to the differences in assessment methods for populations less than or greater than 75,000. Given that the rationale for using either the arithmetic mean or the geometric mean is equivocal, *abundance* metric statuses were presented with equal consideration for both averaging methods.

For *trends in abundance* metrics, considerable work went into reconciling escapement time series for the inclusion in a CUs escapement record. This was a critical step that required a detailed understanding of the history of enumeration sites. The absence of this detailed review of escapement sites would have resulted in biased *trends in abundance* status. For *trends in abundance* metrics, the current paper attempts to address the complexity of the Red, Amber and Green zones for WSP status by presenting the actual metric values and shades of these zones (depending on how close or far to the benchmarks CU values fell) (Table 5). This approach provides more information on the actual CU values, rather than simply presenting one of three colors for each metric. For a large number of CUs, long-term *trends in abundance* were in the Green zone (close to or above the long-term average abundance) and, in contrast, recent trends in abundance in the last three generations, were in the Red zone (Table 5). For most of these CUs, the recent declining trend was largely attributed to the fact that abundances were returning to average after a period of above average abundance in the 1990s. This metric is consistently used by the World Conservation Union (IUCN) and the COSEWIC to determine status. However, the case of Fraser Sockeye emphasizes the importance of placing this metric in the context of the historical time series.

The use of four year running geometric ($\log_e$ transformed) means in *trends in abundance* analyses can produce trends that reflect changes in cyclic abundance rather than actual changes in abundance. Shuswap-ES is an example of a CU that has exhibited a decreasing trend in the last three generations (placing this CU on this metric in the WSP Red zone) that can be attributed to changes in cyclic abundance over time rather than changes in abundance. The changes in the generational geometric mean (used in *trends in abundance* status evaluations) for this CU reflect a shift from a subtle cyclic abundance pattern at the start of the three-generation period to the more typical, highly cyclic pattern of abundance in the most recent generation. Since geometric means downweight dominant cycle years, relative to weak cycle years, geometric means tend to be much lower than arithmetic means for highly cyclic CUs. However, for periods in a CUs time series (or across CUs) where abundance does not exhibit strong cyclical abundance, geometric means will be similar to arithmetic means. In the case of Shuswap-ES, at the start of the three-generation trend period, the CU's dominant cycle was smaller, and weak cycles were larger than the typical (more highly cyclic) years that followed. Therefore, the higher geometric mean at the start of the three-generation trend times series, and the lower geometric mean at the end of this time series, reflects the changes in cyclicity of the CU. When arithmetic means are compared over this same period, this CU increases in abundance rather than decreases.

This paper estimated benchmarks and evaluated statuses for *abundance* and *trends in abundance* indicators (Figure 2). *Distribution* indicators were not considered in the current paper for the several reasons. First, most Fraser Sockeye CUs are lake-type CUs, with individual lakes being the unit of classification for a CU. *Distribution* indicators may not be as necessary for Fraser Sockeye CUs that occupy a smaller freshwater geographic area, relative to more broadly distributed non-Sockeye CUs that are organized by broader joint adaptive zones described by Holtby and Ciruna (2007). In addition, *distributional* changes will be a challenge to assess for Fraser Sockeye due to data collection methods that often preclude the ability to track true distributional trends through time, other than on a coarse scale for most systems. If *distribution* indicators of status are to be used in the future for Fraser Sockeye, they will require considerable input from the programs currently monitoring Sockeye escapements in the Fraser watershed, and will also require linkages to habitat indicators. *Fishing mortality* indicators were the fourth indicator proposed for status evaluation by Holt et al. (2009) (Figure 2). However, these indicators also were not considered in the current paper since they are not intrinsic properties of the CU and generally are not required when *abundance* indicator information is available, such is the case for Fraser Sockeye. The *Fishing mortality* indicator, in fact, might be more appropriate for the characterization of threat to a CU, rather than for the assessment of status. Further, for both *distribution* and *fishing mortality* indicators, appropriate benchmarks have not yet been developed, and are the subject of on-going research.

This paper characterizes uncertainty in *abundance* benchmarks and revises *trends in abundance* benchmarks. Statuses across the range of benchmarks developed for each assessable CU are also presented. Typically, in other assessments of status, benchmarks are deterministic (single point) estimates that do not incorporate uncertainties associated with the assumptions underlying the model form used (structural uncertainty) or uncertainties in how the model fits the existing stock-recruitment data (stochastic uncertainty). The current paper, however, presents uncertainty in benchmarks and consequent status evaluations. These results will provide the necessary foundation for a status aggregation process to develop a final single status for each current and *de novo* (Seton-L) Fraser Sockeye CU. This work will be part of subsequent processes and publications.

NEXT STEPS

- Develop methods for integrating status assessments of each CU across benchmarks and metrics to the extent possible, given contradictory information on status (Red, Amber, or Green) within some CUs. Future workshops and publications are expected as outcomes of this next step.
- Investigate the quantitative properties of benchmarks derived from the Larkin model to determine whether Larkin benchmarks (averaged across cycle lines or cycle-line specific) should be considered in assessment of status on abundances in some or all CUs that exhibit cyclic dynamics.
- Evaluate whether or not *distribution* indicators are required for Fraser Sockeye status assessments. If statuses based on *distribution* indicators are deemed appropriate for these CUs, then modification of existing escapement enumeration study designs to meet the additional objective of assessing Fraser Sockeye distributional changes through time may be required. Development of appropriate benchmarks for *distribution* metrics will concurrently be required.
- To evaluate *trends in abundance* metrics in this paper, considerable effort went into organizing the data, determining which sites to include or exclude, and gap filling the time series'. Similar efforts went into the production database by the PSC. This type of work required considerable input from various experts on the Fraser Sockeye enumeration programs through time, and cannot be done independent of this type of input. This paper attempts to provide the first steps in documenting the current CU escapement data. A

process for providing these time series' in the publicly available NuSeds escapement database was recommended during the November 15/16 2011 review of this paper. The revised time series' would not replace existing escapement time series', but would be additional resources for those without expert knowledge who require this level of data treatment for analysis.

- The current paper updates lake rearing capacity estimates of spawner abundances that maximize juvenile production, for use as informative priors ('*b*' parameter priors of the Ricker model) in the *abundance* benchmark estimation process. However, for some CUs, instead of lake rearing capacity, spawning ground capacity (habitat availability for egg deposition and incubation) may be in fact what is limiting juvenile production. For example, river-type CUs do not rear as juveniles in freshwater lakes, and therefore, will be limited almost entirely by spawning ground capacity in freshwater. Other CUs may have relatively small areas of spawning habitat relative to much more abundant lake rearing habitat, and therefore, juvenile production may again be more limited by spawning capacity instead of lake rearing. Therefore, reliable estimates of spawning ground capacity are required to assist with developing informative priors for all Fraser Sockeye CUs with stock-recruitment data. These data could be used to provide more reliable carrying capacity for CUs with stock-recruitment data. Alternatively, these data could be used to provide a method to estimate *abundance* benchmarks for CUs without stock-recruitment data (Nahatlach-ES, Taseko-ES, Harrison (D/S)-L and Widgeon) or to independently assess the Chilko-ES component of the Chilko-ES/Chilko-S aggregate.

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Table 4. Abundance metric statuses for each assessable Fraser Sockeye Conservation Unit (CU) are presented across a range of model forms (structural uncertainty), probability levels (stochastic uncertainty), and methods for averaging recent abundances (geometric versus arithmetic). For each CU, benchmarks are presented for full time series standard Ricker models (first model in the series) and truncated Ricker models (second and possibly third model in the series, depending on the CU), with the time series length indicated in brackets besides the model name. The final two model forms used to estimate benchmarks for Fraser Sockeye, include the smoothed-Ricker and the recursive-Bayesian Ricker models. For each CU and model form, the cumulative probability distributions (from the 10% to 90% probability levels) of lower and upper benchmarks are also presented. Abundance metric status is evaluated by comparing a CU's arithmetic and geometric mean of the last generation ETS (2006-2009) in relation to these benchmarks. WSP Status is Red if the last generation ETS is below the lower benchmark, Green if it is above the upper benchmark and Amber if it is between the lower and upper benchmarks.

		Abundance (Effective Total Spawner) Lower Benchmark					Abundance (Effective Total Spawner) Upper Benchmark					(ETS) Arithmetic Mean	Stock Status					(ETS) Geometric Mean	Stock Status				
Run Timing Group Conservation Unit	model (time series)	10%	25%	50%	75%	90%	10%	25%	50%	75%	90%	Last Gen. 2006-2009	Probability Level 10%	25%	50%	75%	90%	Last Gen. 2006-2009	Probability Level 10%	25%	50%	75%	90%
Early Stuart Run																							
Takla-Trembleur-EStu	**Ricker (1950-2004)**	46,000	56,000	68,000	86,000	111,000	174,000	192,000	218,000	259,000	302,000	**26,500**	■	■	■	■	■	**20,200**	■	■	■	■	■
	Ricker (1970-2004)	63,000	76,000	97,000	127,000	154,000	177,000	198,000	225,000	263,000	298,000		■	■	■	■	■		■	■	■	■	■
	Ricker (1990-2004)	85,000	95,000	111,000	126,000	130,000	138,000	138,000	143,000	147,000	147,000		■	■	■	■	■		■	■	■	■	■
	smoothed-Ricker (1950-2004)	62,000	72,000	85,000	104,000	126,000	74,000	84,000	97,000	117,000	139,000		■	■	■	■	■		■	■	■	■	■
	Recursive-Bayes Ricker (1950-2004)	82,000	107,000	138,000	176,000	218,000	83,000	131,000	187,000	261,000	332,000		■	■	■	■	■		■	■	■	■	■
Early Summer Run																							
Bowron-ES	**Ricker (1950-2004)**	3,000	3,000	4,000	5,000	6,000	13,000	15,000	17,000	19,000	22,000	**1,600**	■	■	■	■	■	**1,500**	■	■	■	■	■
	Ricker (1970-2004)	3,000	3,000	4,000	5,000	7,000	11,000	13,000	14,000	17,000	18,000		■	■	■	■	■		■	■	■	■	■
	Ricker (1990-2004)	3,000	4,000	5,000	7,000	9,000	10,000	11,000	12,000	14,000	15,000		■	■	■	■	■		■	■	■	■	■
	smoothed-Ricker (1950-2004)	4,000	5,000	5,000	6,000	7,000	5,000	6,000	6,000	7,000	8,000		■	■	■	■	■		■	■	■	■	■
Kamloops-ES	**Ricker (1950-2004)**	3,000	4,000	6,000	9,000	15,000	15,000	18,000	23,000	33,000	50,000	**9,500**	□	□	□	□	■	**9,000**	□	□	□	□	■
	Ricker (1970-2004)	3,000	5,000	7,000	11,000	18,000	15,000	19,000	25,000	34,000	53,000		□	□	□	■	■		□	□	□	■	■
	Ricker (1990-2004)	2,000	3,000	4,000	8,000	14,000	14,000	17,000	22,000	32,000	48,000		□	□	□	□	■		□	□	□	□	■
	Recursive-Bayes Ricker (1950-2004)	5,000	8,000	19,000	44,000	64,000	20,000	30,000	60,000	128,000	188,000		□	□	■	■	■		□	□	■	■	■
Anderson-Seton-ES	**Ricker (1968-2004)**	1,000	2,000	3,000	4,000	9,000	12,000	14,000	19,000	27,000	46,000	**4,100**	□	□	□	□	■	**3,300**	□	□	□	■	■
	Ricker (1990-2004)	2,000	3,000	6,000	14,000	33,000	12,000	16,000	24,000	46,000	88,000		□	□	■	■	■		□	□	■	■	■
	smoothed-Ricker (1968-2004)	2,000	3,000	5,000	7,000	11,000	9,000	11,000	14,000	20,000	31,000		□	□	■	■	■		□	□	■	■	■
	Recursive-Bayes Ricker (1968-2004)[1]	15,000	34,000	70,000	112,000	143,000	45,000	84,000	188,000	325,000	419,000		■	■	■	■	■		■	■	■	■	■
Nadina-Francois-ES (new mixed CU)	**Ricker (1973-2004)**	8,000	11,000	17,000	33,000	59,000	35,000	42,000	58,000	100,000	158,000	**9,400**	□	■	■	■	■	**7,000**	■	■	■	■	■
	Ricker (1990-2004)	7,000	11,000	18,000	40,000	76,000	29,000	36,000	48,000	84,000	121,000		□	■	■	■	■		□	■	■	■	■
	smoothed-Ricker (1973-2004)	10,000	13,000	20,000	33,000	61,000	27,000	34,000	48,000	75,000	132,000		■	■	■	■	■		■	■	■	■	■
	Recursive-Bayes Ricker (1973-2004)[1]	39,000	50,000	80,000	131,000	180,000	71,000	118,000	203,000	305,000	382,000		■	■	■	■	■		■	■	■	■	■
North Barriere-ES (*de novo*)	**Ricker (1958-2004)**	310	380	510	660	820	4,100	4,500	5,000	5,500	6,200	**5,900**	■	■	■	■	□	**2,800**	□	□	□	□	□
	Ricker (1990-2004)	440	610	940	1,500	2,200	4,300	4,800	5,600	6,800	8,000		■	■	■	□	□		□	□	□	□	□
	Recursive-Bayes Ricker (1950-2004)	1,000	1,000	2,000	2,000	3,000	5,000	6,000	7,000	8,000	10,000		■	□	□	□	□		□	□	□	□	■

[1] For these models the Smax estimated is unrealistically high relative to those estimated with all other models, therefore these models are not recommended for benchmark considerations

Table 4. Continued (see previous page for description)

Run Timing Group		Abundance (Effective Total Spawner) Lower Benchmark					Abundance (Effective Total Spawner) Upper Benchmark					Abundance (Effective Total Spawner) Arithmetic	Stock Status Probability Level					Abundance (Effective Total Spawner) Geometric	Stock Status Probability Level				
Conservation Unit	model (time series)	10%	25%	50%	75%	90%	10%	25%	50%	75%	90%	2006-2009	10%	25%	50%	75%	90%	2006-2009	10%	25%	50%	75%	90%
Early Summer Run Continued																							
Pitt-ES	Ricker (1954-2004)	4.000	5.000	6.000	8.000	9.000	16.000	20.000	22.000	24.000	26.000	32,200	■	■	■	■	■	29,300	■	■	■	■	■
	Ricker (1976-2004)	5.000	6.000	7.000	9.000	12.000	18.000	20.000	22.000	24.000	27.000		■	■	■	■	■		■	■	■	■	■
	Ricker (1990-2004)	3.000	4.000	6.000	8.000	12.000	19.000	20.000	22.000	24.000	27.000		■	■	■	■	■		■	■	■	■	■
	smoothed-Ricker (1950-2004)	6.000	8.000	9.000	11.000	11.000	14.000	16.000	17.000	18.000	19.000		■	■	■	■	■		■	■	■	■	■
	Recursive-Bayes Ricker (1939-2004)	5.000	8.000	10.000	13.000	15.000	20.000	24.000	27.000	31.000	35.000		■	■	■	■	▨		■	■	■	▨	▨
Shuswap-ES	Ricker (1953-2004)	37.000	56.000	88.000	156.000	253.000	113.000	144.000	188.000	304.000	437.000	64,600	▨	▨	■	■	■	13,500	■	■	■	■	■
	Ricker (1995-2004)	34.000	54.000	87.000	161.000	238.000	91.000	119.000	155.000	241.000	308.000		▨	▨	■	■	■		■	■	■	■	■
	smoothed-Ricker (1948-2004)	27.000	36.000	51.000	79.000	130.000	71.000	87.000	113.000	161.000	248.000		▨	▨	▨	■	■		■	■	■	■	■
	Recursive-Bayes Ricker (1948-2004)[1]	76.000	109.000	200.000	285.000	391.000	184.000	287.000	387.000	571.000	714.000		■	■	■	■	■		■	■	■	■	■
Chilliwack-ES	Carrying Capacity of system (deterministic, not probabilistic)	–	–	8.000	–	–	–	–	16.000	–	–	12,000	□	□	▨	□	□	6,000	□	□	■	□	□
Summer Run																							
Chilko-S & Chilko-ES aggregate	Ricker (1953-2004)	20.000	33.000	39.000	47.000	54.000	236.000	252.000	273.000	284.000	311.000	278,000	■	■	■	▨	▨	248,700	■	▨	▨	▨	▨
	Ricker (1976-2004)	20.000	25.000	31.000	39.000	50.000	215.000	235.000	252.000	274.000	304.000		■	■	■	■	▨		■	■	▨	▨	▨
	Ricker (1990-2004)	21.000	29.000	43.000	66.000	92.000	216.000	239.000	258.000	285.000	310.000		■	■	■	▨	▨		■	■	▨	▨	▨
	smoothed-Ricker (1950-2004)	44.000	51.000	61.000	72.000	85.000	200.000	208.000	222.000	236.000	253.000		■	■	■	■	■		■	■	■	■	▨
	Recursive-Bayes Ricker (1958-2004)	37.000	46.000	63.000	81.000	96.000	197.000	223.000	250.000	270.000	309.000		■	■	■	▨	▨		■	■	▨	▨	▨
Takla-Trembleur-Stuart-S	Ricker (1950-2004)	55.000	73.000	104.000	146.000	187.000	343.000	400.000	489.000	606.000	741.000	59,100	▨	■	■	■	■	38,400	■	■	■	■	■
	Ricker (1975-2004)	64.000	87.000	125.000	186.000	267.000	365.000	414.000	500.000	622.000	763.000		■	■	■	■	■		■	■	■	■	■
	Ricker (1990-2004)	129.000	170.000	234.000	309.000	392.300	349.000	390.000	444.000	489.000	549.000		■	■	■	■	■		■	■	■	■	■
	smoothed-Ricker (1950-2004)	105.370	150.000	209.000	284.000	382.500	216.000	336.000	482.000	703.000	958.000		■	■	■	■	■		■	■	■	■	■
	Recursive-Bayes Ricker (1950-2004)	110.000	134.000	168.000	210.000	272.300	164.000	182.000	231.000	284.000	359.000		■	■	■	■	■		■	■	■	■	■
Quesnel-S	Ricker (1950-2004)	84.000	100.000	121.000	140.000	168.800	600.000	637.000	701.000	742.000	805.000	95,800	▨	■	■	■	■	61,500	■	■	■	■	■
	Ricker (1975-2004)	76.000	91.000	112.000	139.000	164.300	615.000	647.000	684.000	736.000	775.000		▨	▨	■	■	■		■	■	■	■	■
	Ricker (1990-2004)	126.000	160.000	202.000	253.000	291.300	516.000	529.000	539.000	530.000	521.000		■	■	■	■	■		■	■	■	■	■
	smoothed-Ricker (1953-2004)	107.000	200.000	234.000	253.000	268.500	344.000	366.000	380.000	376.000	384.000		■	■	■	■	■		■	■	■	■	■
	Recursive-Bayes Ricker (1950-2004)	159.000	213.000	255.000	294.000	331.300	165.000	322.000	464.000	580.000	693.000		■	■	■	■	■		■	■	■	■	■
Francois-Fraser-S	Ricker (1950-2004)	27.000	33.000	42.000	53.000	68.000	151.000	166.000	195.000	225.000	264.000	87,500	▨	▨	▨	▨	▨	68,000	▨	▨	▨	▨	▨
	Ricker (1975-2004)	29.000	34.000	43.000	56.000	76.000	152.000	170.000	192.000	238.000	274.000		▨	▨	▨	▨	▨		▨	▨	▨	▨	■
	Ricker (1990-2004)	44.000	57.000	79.000	102.000	130.000	158.000	172.000	197.300	219.000	246.000		▨	▨	▨	■	■		▨	▨	■	■	■
	smoothed-Ricker (1950-2004)	43.000	56.000	67.000	79.000	90.000	76.000	84.000	90.000	98.000	107.000		■	■	▨	▨	■		▨	▨	▨	■	■
	Recursive-Bayes Ricker (1950-2004)	65.000	79.000	96.000	117.000	140.000	95.800	129.000	164.300	205.000	249.000		▨	▨	■	■	■		▨	■	■	■	■

1. For those models the Smax estimated is unrealistically high relative to those estimated with all other models, therefore these models are not recommended for benchmark considerations

Table 4. Continued (see previous page for description)

Run Timing Group Conservation Unit	model (time series)	Abundance (Effective Total Spawner) Lower Benchmark					Abundance (Effective Total Spawner) Upper Benchmark					Abundance (Effective Total Spawner) Arithmetic	Abundance (Effective Total Spawner) Stock Status					Abundance (Effective Total Spawner) Geometric	Abundance (Effective Total Spawner) Stock Status				
		10%	25%	50%	75%	90%	10%	25%	50%	75%	90%	Last Gen. 2006-2009	Probability Level 10%	25%	50%	75%	90%	Last Gen. 2006-2009	Probability Level 10%	25%	50%	75%	90%
Late Run																							
Cultus-L	Ricker ([illegible])	9,000	10,000	12,000	15,000	17,000	26,000	28,000	32,000	34,000	36,000	900	■	■	■	■	■	600	■	■	■	■	■
	Ricker ([illegible])	8,000	10,000	12,000	15,000	16,000	27,000	29,000	32,000	34,000	36,000		■	■	■	■	■		■	■	■	■	■
	Ricker ([illegible])	11,000	12,000	[illegible]	13,000	8,000	[illegible]	18,000	[illegible]	[illegible]	7,000		■	■	■	■	■		■	■	■	■	■
	smoothed-Ricker ([illegible])	11,000	12,000	14,000	16,000	18,000	16,000	18,000	21,000	24,000	26,000		■	■	■	■	■		■	■	■	■	■
	Recursive-Bayes Ricker ([illegible])	8,000	11,000	13,000	16,000	18,000	7,000	15,000	22,000	30,000	38,000		■	■	■	■	■		■	■	■	■	■
Harrison River (River-Type)	Ricker ([illegible])	6,000	7,000	8,000	11,000	14,000	28,000	30,000	32,000	38,000	40,000	147,700	■	■	■	■	■	80,300	■	■	■	■	■
	Ricker (1975-2008)	6,000	7,000	9,000	11,000	14,000	33,000	38,000	38,000	44,000	48,000		■	■	■	■	■		■	■	■	■	■
	Ricker ([illegible])	3,000	4,000	6,000	8,000	13,000	34,000	38,000	43,000	49,000	58,000		■	■	■	■	■		■	■	■	■	■
	smoothed-Ricker ([illegible])	100	100	200	300	300	7,000	8,000	10,000	11,000	13,000		■	■	■	■	■		■	■	■	■	■
	Recursive-Bayes Ricker ([illegible])[1]	3,000	3,000	3,000	3,000	3,000	5,000	6,000	7,000	8,000	11,000		■	■	■	■	■		■	■	■	■	■
Shuswap Complex-L	Ricker (1950-2004)	234,000	263,000	305,000	434,000	546,000	1,070,000	1,162,000	1,280,000	1,415,000	1,633,000	578,400	□	□	□	□	□	28,900	■	■	■	■	■
	Ricker (1970-2004)	301,000	268,000	337,000	447,000	558,000	894,000	1,090,000	1,194,000	1,384,000	1,593,000		□	□	□	□	□		■	■	■	■	■
	Ricker (1990-2004)	171,000	238,000	338,000	460,000	576,000	707,000	836,000	879,000	962,000	915,000		□	□	□	□	□		■	■	■	■	■
	smoothed-Ricker (1950-2004)	346,000	364,000	376,000	383,000	394,000	347,000	373,000	384,000	400,000	410,000		■	■	■	■	■		■	■	■	■	■
	Recursive-Bayes Ricker (1950-2004)	330,000	417,000	510,000	632,000	768,000	525,000	870,000	1,166,000	1,432,000	1,728,000		■	□	□	■	■		■	■	■	■	■
Seton-L (de novo)	Ricker ([illegible])	900	1,000	1,000	1,000	2,000	6,000	7,000	8,000	10,000	12,000	5,300	□	□	□	□	□	1,400	□	□	□	□	■
	Ricker ([illegible])	1,000	1,000	2,000	4,000	7,000	6,000	7,000	10,000	14,000	23,000		□	□	□	□	■		□	□	■	■	■
	smoothed-Ricker ([illegible])	1,000	1,000	2,000	2,000	3,000	5,000	5,000	6,000	7,000	9,000		■	■	□	□	□		□	□	■	■	■
	Recursive-Bayes Ricker ([illegible])	1,000	1,000	2,000	2,000	3,000	5,000	5,000	6,000	7,000	9,000		■	■	□	□	□		□	□	■	■	■
Harrison (U/S)-L	Ricker ([illegible])	4,000	6,000	9,000	14,000	23,000	52,000	61,000	76,000	103,000	147,000	20,400	□	□	□	□	■	12,200	□	□	□	■	■
	Ricker ([illegible])	1,000	2,000	4,000	7,000	13,000	25,000	32,000	43,000	63,000	89,000		□	□	□	□	□		□	□	□	□	■
	smoothed-Ricker ([illegible])	7,000	9,000	13,000	19,000	30,000	45,000	54,000	67,000	90,000	133,000		□	□	□	□	■		□	□	■	■	■
	Recursive-Bayes Ricker ([illegible])[1]	11,000	16,000	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]		□	□	■	■	■		□	■	■	■	■
Lillooet-Harrison-L	Ricker ([illegible])	6,000	9,000	11,000	14,000	17,000	67,000	72,000	77,000	85,000	93,000	104,900	■	■	■	■	■	63,600	□	□	□	□	□
	Ricker ([illegible])	7,000	9,000	13,000	17,000	21,000	67,000	73,000	80,000	88,000	94,000		■	■	■	■	■		□	□	□	□	□
	Ricker ([illegible])	15,000	20,000	26,000	33,000	39,000	27,000	59,000	81,000	89,000	95,000		■	■	■	■	■		■	■	■	■	■
	smoothed-Ricker ([illegible])	15,000	18,000	22,000	26,000	30,000	48,000	50,000	51,000	52,000	54,000		■	■	■	■	■		■	■	■	■	■
	Recursive-Bayes Ricker ([illegible])	17,000	22,000	27,000	31,000	38,000	28,000	40,000	55,000	68,000	78,000		■	■	■	■	■		■	■	■	□	□

1. For these models the Smax estimated is unrealistically high relative to those estimated with all other models; therefore these models are not recommended for benchmark considerations

Table 5. Trends in abundance statuses for each of the 26 assessable CUs. To interpret trends in abundance statuses, the long-term (1950-2009) average effective female spawner (EFS) abundances, average EFS in the third generation from the end of the time series (Gen. 3rd from Last), and the last generation average EFS (Last Gen.) are provided (note: different (transformed) values are used in the actual status assessments). For the long-term trends in abundance metric, the ratio of the current generation EFS geometric mean to the historical geometric EFS mean is presented for each CU, and colour coded to correspond with status. Statuses are assessed by comparing this ratio to the upper benchmark (UB) of 0.75 and lower benchmark (LB) of 0.5. For the recent trends in abundance metric, the linear change in abundance over the last three generations is presented for each CU and colour coded to correspond with status. Status is assessed by comparing this abundance change to the LB of a 25% reduction and the UB of a 15% reduction. The second recent trends in abundance metric is the probability that this rate of change is below the lower benchmark of 25%, and it is complementary to the first, deterministic recent trends in abundance metric. Statuses are color coded Red if they are below the lower benchmark, Green if they are above the upper benchmark and Amber if they are between the LB and UB for each metric. Data quality for trends in abundance and abundance metrics are presented in final column (codes described in preceding section on Data Quality).

	Escapement: Effective Female Spawner (EFS) Averages			**Long-Term Trends (EFS)** (Metric 1)	**Recent Trends (EFS)** (Metric 2)	(Metric 3)	
Run Timing Group Conservation Unit	**Long-Term Average**	**Gen. 3rd from Last**	**Last Gen.**	**Ratio of Current Generation to Historical Average**	**Linear Change in Abundance in the Last Three Generations**	**Prob. Decline <LB**	**Data Quality**
				0.25 0.50 0.63 0.75 1.0	-50% -25% -15% +%		
Early Stuart							
Takla-Trembleur-EStu	40,900	31,000	13,300	0.58	[illegible]	1.00	4
Early Summer							
Bowron-ES	4,300	3,900	800	0.27	[illegible]	0.99	3
Kamloops-ES	4,200	13,000	5,300	[illegible]	[illegible]	0.12	3
Anderson-Seton-ES	3,600	6,200	2,400	[illegible]	-38%	0.80	3
Nadina-Francois-ES (new mixed CU)	9,000	22,600	4,800	0.96	-37%	0.69	3
North Barriere-ES (*de novo*)	3,800	4,100	3,900	[illegible]	[illegible]	1.00	3
Pitt-ES	13,200	38,900	15,800	[illegible]	0%	0.27	4
Shuswap-ES	19,900	15,600	36,800	0.9	-38%	0.73	4

Table 5. Continued (see previous pages for description).

	Escapement: Effective Female Spawner (EFS) Averages			**Long-Term Trends (EFS)** (Metric 1) Ratio of Current Generation to Historical Average						**Recent Trends (EFS)** (Metric 2) Linear Change in Abundance in the Last Three Generations					(Metric 3)	
Run Timing Group Conservation Unit	Long-Term (1950-2009)	Gen. 3rd from Last	Last Gen.		0.25	0.50	0.63	0.75	1.0		-50%	-25%	-15%	+%	Prob. Decline <LB	Data Quality
				■	▨	□	□	□	■	■	▨	□	□	■		
Early Summer Continued																
Nahatlach-ES	2,200	2,300	1,000			0.55				■					1.00	3
Chilliwack-ES[1]	1,100	1,400	500	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	3
Taseko-ES	1,300	800	400		0.32					■					0.97	2
Summer																
Chilko-S & Chilko-ES	191,600	406,800	153,600						[illegible]	■					1.00	4
Takla-Trembleur-Stuart-S	76,700	122,400	28,100						[illegible]	■					0.99	4
Quesnel-S	188,700	585,600	50,700						[illegible]	■					0.99	4
Francois-Fraser-S	53,000	105,000	47,300						[illegible]		-38%				0.78	4
Late																
Cultus-L[2]	11,800	1,100	800	■						■					1.00	5
Harrison River (River Type)	13,600	4,700	63,400						[illegible]					[illegible]	0.00	3
Shuswap Complex-L	312,300	204,300	303,700					0.95						[illegible]	0.12	4
Seton-L (de novo CU)	3,800	4,100	3,200					0.91		■					0.95	3
Harrison (U/S)-L	19,200	13,400	10,700					0.8			-30%				0.60	3
Harrison (D/S)-L	1,500	3,200	4,300						[illegible]					[illegible]	0.01	3
Lillooet-Harrison-L	44,200	59,000	58,200						[illegible]					[illegible]	0.02	4
Widgeon (River-Type)	300	30	200		0.35									[illegible]	0.00	3

1. Chilliwack-ES cannot be assessed quantitatively due to the short time series for Dolly Varden Creek.
2. Cultus is effective total wild spawners since sex identification at the fence during enumeration is a challenge

APPENDIX 1: For each conservation unit, the associated Sockeye enumeration sites that occur in the escapement database are indicated, and a checkmark beside the site name indicates its use in the escapement time series to evaluate CU status for *Trends in Abundance* metrics.

Anderson-Seton-ES

Sites	IN
Gates Channel	√
Gates Creek	√

Bowron-ES

Sites	IN
Antler Creek	
Bowron River	√
Pomeroy Creek	√
Huckey Creek	√
Sus Creek	√

Chilko-S & Chilko-ES

Sites	IN
Chilko River	√
Chilko Channel	√
Chilko Lake North	√
Chilko Lake South	√

Chilliwack-ES

Sites	IN
Chilliwack Lake	√
Dolly Varden Creek	√

Cultus-L

Sites	IN
Cultus Lake	√

Francois-Fraser-S

Sites	IN
Stellako River	√
Uncha Creek	
Ormonde Creek	

Harrison (D/S)-L

Sites	IN
Bear Creek	
Big Silver Creek	√
Cogburn Creek	
Crazy Creek	
Douglas Creek	
Hatchery Creek	
Sloquet Creek	
Tipella Creek	
Tipella Slough	

Harrison (U/S)-L

Sites	IN
East Creek	√
Weaver Channel	√
Weaver Creek	√

Harrison River (River-Type)

Sites	IN
Harrison River	√

Kamloops-ES

Sites	IN
Barriere River	
Clearwater River	
Dunn Creek	
Finn Creek	
Grouse Creek	
Hemp Creek	
Lemieux Creek	
Lion Creek	
Mann Creek	
Moul Creek	
North Thompson River	
Raft River	√

Lilloet-Harrison-L

Sites	IN
Birkenhead River	√
Green River	
Lillooet Slough	
Miller Creek	
Poole Creek	
Railroad Creek	
Ryan Creek	
Sampson Creek	
John Sandy (not in database)	
25 Mile Creek	

Nadina-Francois-ES (new mixed CU)

Sites	IN
Glacier Creek	
Early Nadina River	√
Late Nadina River	√
Nadina Channel	√
Tagetochlain Creek	

Nahatlach-ES

Sites	IN
Nahatlach Lake	√
Nahatlach River	√

North Barriere-ES (*de novo*)

Sites	IN
Fennell Creek	√
Harper Creek	

APPENDIX 1. Continued (see previous page for description).

Pitt-ES		Quesnel-S						Seton-L *(de novo)*	
Sites	IN	Sites	IN	Sites	IN	Sites	IN	Sites	IN
Upper Pitt River	√	Abbott Creek		Isaiah Creek		Tisdall Creek		Portage Creek	√
		Amos Creek		Junction Creek		Trickle Creek			
		Archie Creek		Junction Creek - shore		Wasko Creek			
		Baxter Beach		Killdog Creek		Wasko Creek - shore			
		Bear Beach - shore		Killdog Creek - shore		Watt Creek			
		Betty Frank's - shore		Lester Shore		Watt Creek - shore			
		Big Slide - shore		Limestone Creek		Whiffle Creek			
		Big Slide - shore 1km West		Limestone Point - shore		Winkley Creek			
		Bill Miner Creek		Limestone Point - shore 5km South					
		Bill Miner Creek - shore		Little Horsefly River	√				
		Bill Miner Creek - shore 3km West		Logger Landing					
		Blue Lead Creek		Long Creek					
		Blue Lead Creek - shore		Long Creek - shore					
		Bouldery Creek		Lynx Creek					
		Bouldery Creek - shore		Lynx Creek - shore					
		Bouldery Creek - shore 2km East		Marten Creek					
		Bowling Point		Marten Creek - shore					
		Buckingham Creek		McKinley Creek	√				
		Cameron Creek	√	McKinley Creek - Lower	√				
		Clearbrook Creek		McKinley Creek - Upper	√				
		Deception Point		Mitchell River	√				
		Devoe Creek		Moffat Creek					
		Devoe Creek - shore		Niagara Creek					
		Double T - shore		North Arm - shore (Bowling-Goose Pt.)					
		East Arm - shore (Rock Slide-Penninsula Pt)		North Arm - shore (Roaring-Deception Pt.)					
		East Arm - unnamed creek 1		North Arm - unnamed cove					
		East Arm - unnamed creek 2 - shore		Opa Beach					
		East Arm - unnamed point		Penfold Camp Shore					
		Elysia - shore		Penfold Creek	√				
		Elysia - shore 1km West		Quartz Point					
		Franks Creek		Quesnel Lake					
		Franks Creek - shore		Raft Creek					
		Goose Creek		Roaring Point					
		Goose Point - shore		Roaring River					
		Goose Point - shore 8km South		Roaring River - shore					
		Grain Creek		Rock Slide					
		Grain Creek - shore		Service Creek					
		Hazeltine Creek		Slate Bay					
		Horsefly Channel	√	Slate Bay 1km East					
		Horsefly Lake		Spusks Creek					
		Horsefly River	√	Sue Creek					
		Horsefly River - Above Falls	√	Summit Creek					
		Horsefly River - Lower	√	Taku Creek					
		Horsefly River - Upper	√	Tasse Creek					
		Hurricane Point		Tasse Creek - shore					

APPENDIX 1. Continued (see previous page for description).

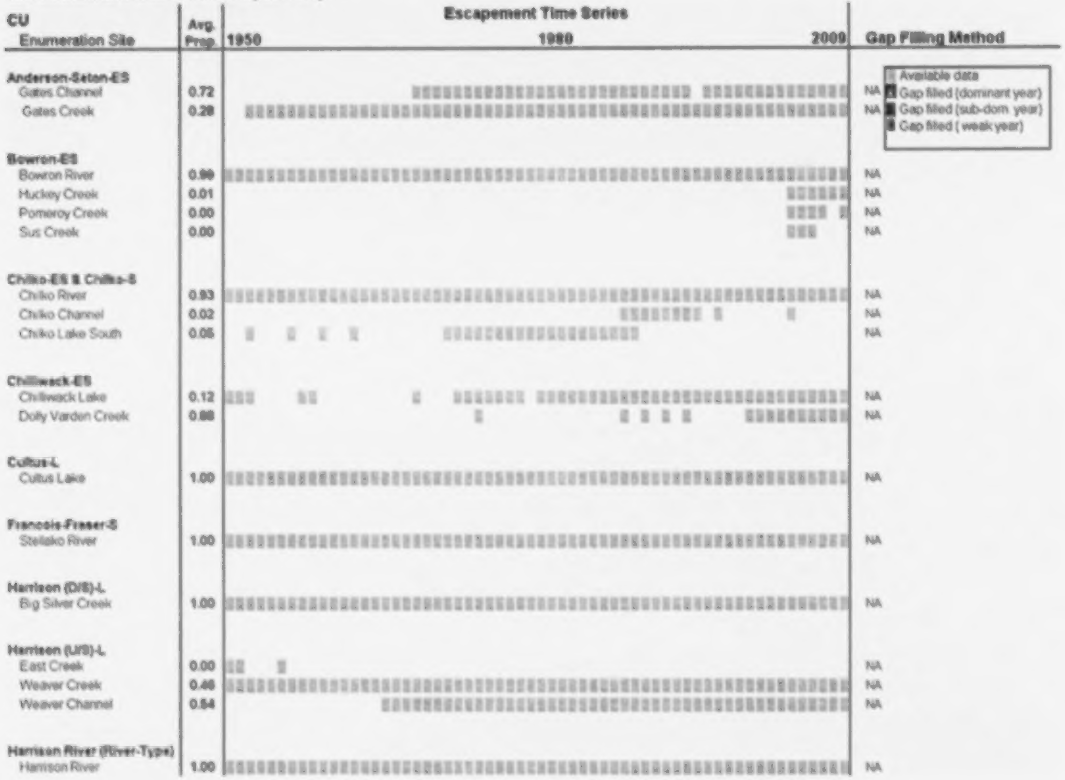
Shuswap-ES		Shuswap Complex-L				Takla-Trembleur-EStu				Takla-Trembleur-Stuart-S	
Sites	IN	Sites	IN	Sites	IN	Sites	IN	Sites	IN	Sites	IN
Adams Channel		5 Mile Creek		Pass Creek	√	5 Mile Creek	√	Sakeniche River	√	Kazchek Creek	√
Adams River		Adams Channel		Pass Creek - shore		10 Mile Creek		Sandpoint Creek	√	Kuzkwa Creek	√
Anstey River		Adams Lake		Perry River		15 Mile Creek	√	Shale Creek	√	Middle River	√
Burton Creek		Adams Lake - East		Queest Creek - shore		25 Mile Creek	√	Sinta Creek	√	Pinchi Creek	
Bush Creek		Adams Lake - North		Reinecker Creek		Ankwill Creek	√	Takla Lake - shore		Sakeniche River	
Celista Creek		Adams Lake - South		Reinecker Creek - shore		Baptiste Creek		Takla Lake - unnamed		Sowchea Creek	
Craigellachie Creek		Adams River	√	Ross Creek		Bates Creek		Tarnezell Creek		Stuart Lake	
Crazy Creek		Adams River - shore	√	Ross Creek - shore	√	Bivouac Creek	√	Tildesley Creek		Stuart River	
Eagle River		Anstey River	√	Salmon Arm-shore		Blackwater Creek	√	Tliti Creek		Tachie River	√
Huihill (Bear) Creek		Anstey River - shore	√	Salmon River		Blanchette Creek		Unnamed Creek			
Hunakwa Creek		Hiuhill (Bear) Creek		Scotch Creek	√	Casamir Creek					
Loftus Creek		Bessette Creek		Scotch Creek - shore	√	Consolidated Creek	√				
McNomee Creek	√	Bush Creek		Seymour River		Crow Creek	√				
Middle Shuswap River		Bush Creek - shore		Shuswap Lake	√	Driftwood River	√				
Nikwikwaia (Gold) Creek		Canoe Creek		Shuswap Lake - Anstey Arm	√	Dust Creek	√				
Onyx Creek		Celista Creek		Shuswap Lake - Main Arm	√	Felix Creek	√				
Pass Creek		Cook Creek		Shuswap Lake - Main Arm North	√	Fleming Creek					
Perry River		Crazy Creek		Shuswap Lake - Main Arm South	√	Forfar Creek	√				
Ross Creek		Cruikshank Pt.W.-shore	√	Shuswap Lake - Salmon Arm	√	Forsythe Creek	√				
Salmon River		Eagle River	√	Shuswap Lake - Salmon Arm East	√	French Creek					
Scotch Creek	√	Four Mile Creek- shore		Shuswap Lake - Salmon Arm North	√	Frypan Creek	√				
Seymour River	√	Nikwikwaia (Gold) Creek		Shuswap Lake - Salmon Arm South	√	Gluske Creek	√				
Yard Creek		Hlina Creek - shore	√	Shuswap Lake - Seymour Arm	√	Hooker Creek					
		Hunakwa Creek		South Thompon River	√	Hudson Bay Creek					
		Kingfisher Creek		Tappen Creek		Kastberg Creek	√				
		Knight Creek - shore		Trinity Creek		Kazchek Creek					
		Lee Creek - shore	√	Tsikwustum Creek		Kotesine Creek	√				
		Little River	√	Tsuius Creek		Kynock Creek	√				
		Loftus Creek		Upper Adams River		Leo Creek					
		Lower Shuswap River	√	Vanishing Creek - shore		Lion Creek	√				
		Mara Lake - shore		Wap Creek		McDougall Creek					
		McNomee Creek		Yard Creek		Middle River (Rosette)					
		Middle Shuswap River	√			Nancut Creek					
		Momich River	√			Narrows Creek	√				
		Momich River - shore				Paula Creek	√				
		Noisy Creek				Point Creek	√				
		Onyx Creek				Porter Creek	√				
		Onyx Creek - shore	√			Rosette Creek	√				

APPENDIX 1. Continued (see previous page for description).

Taseko-ES		Widgeon (River-Type)	
Sites	IN	Sites	IN
Taseko Lake	√	Widgeon Creek	√

APPENDIX 2. The annual escapement (EFS) time series by year (grey filled rectangles) and enumeration site, used for each CU's *trends in abundance* status analyses. This appendix documents when gap filling was required for an enumeration site on a dominant cycle years (black filled rectangles with gold star), sub-dominant cycle years (black filled rectangle) or weak cycle years (blue filled rectangle). The specific gap filling method is reported in the final column (methods presented in Appendix 5). The average proportion (across all years of data) an enumeration site's EFS represents, out of the CUs total EFS, is indicated in the second column.

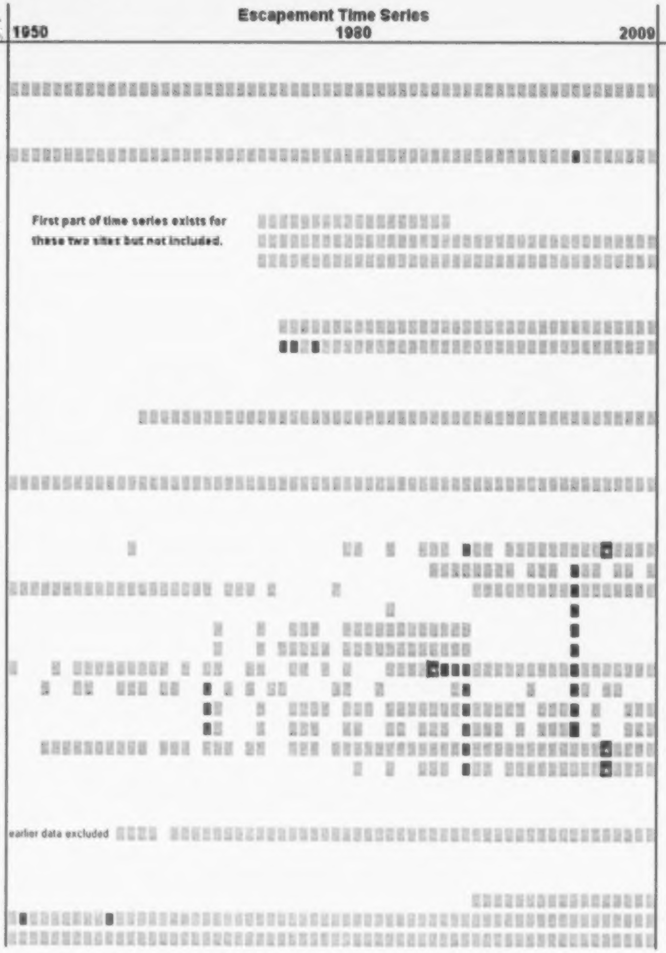
1. Pre- and post-1980 time series separated when calculating proportions
2. Streams grouped according to correlation of trends prior to gap filling
3. Dominant and Sub-dominant cycles only



CU Enumeration Site	Avg. Prop.	Gap Filling Method
Anderson-Seton-ES		
Gates Channel	0.72	NA
Gates Creek	0.28	NA
Bowron-ES		
Bowron River	0.99	NA
Huckey Creek	0.01	NA
Pomeroy Creek	0.00	NA
Sus Creek	0.00	NA
Chilko-ES & Chilko-S		
Chilko River	0.93	NA
Chilko Channel	0.02	NA
Chilko Lake South	0.05	NA
Chilliwack-ES		
Chilliwack Lake	0.12	NA
Dolly Varden Creek	0.88	NA
Cultus-L		
Cultus Lake	1.00	NA
Francois-Fraser-S		
Stellako River	1.00	NA
Harrison (D/S)-L		
Big Silver Creek	1.00	NA
Harrison (U/S)-L		
East Creek	0.00	NA
Weaver Creek	0.46	NA
Weaver Channel	0.54	NA
Harrison River (River-Type)		
Harrison River	1.00	NA

APPENDIX 2. Continued (see previous page for description).

1. Pre- and post-1980 time series separated when calculating proportions
2. Streams grouped according to correlation of trends prior to gap filling
3. Dominant and Sub-dominant cycles only

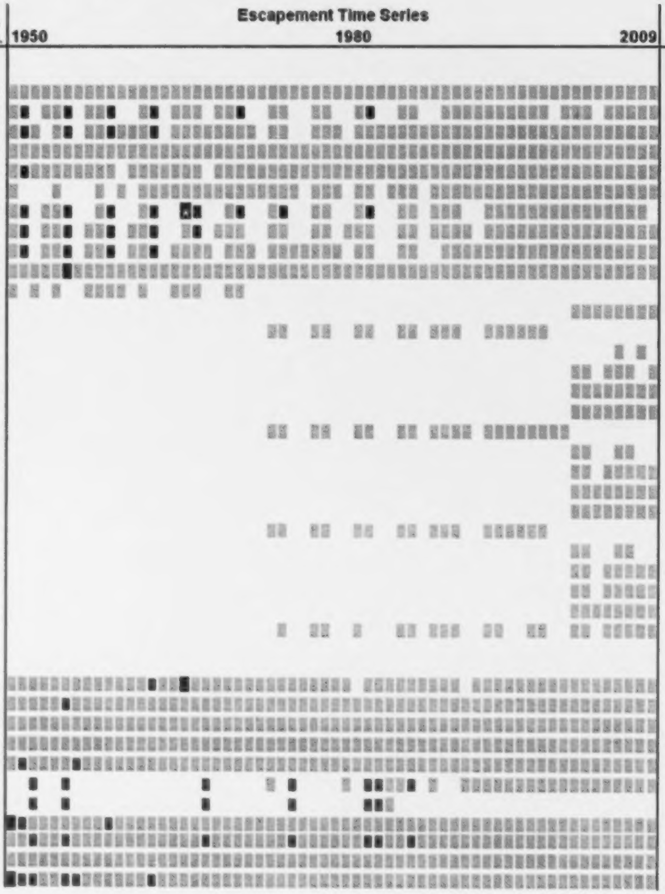


CU Enumeration Site	Avg. Prop.	Gap Filling Method
Kamloops-ES		
Raft River	0.60	NA
Lillooet-Harrison-L		
Birkenhead River	1.00	Cycle-Line Average
Nadina-Francois-ES (mixed)		
Early Nadina River	0.12	NA
Late Nadina River	0.34	NA
Nadina Channel	0.54	NA
Nahatlatch-ES		
Nahatlatch River	0.75	NA
Nahatlatch Lake	0.25	Mean Proportion Method
North Barriere-ES (*de novo*)		
Fennell Creek	0.93	NA
Pitt-ES		
Upper Pitt River	1.00	NA
Quesnel-S		
Cameron Creek	0.03	Mean Proportion Method - Cyclic 1,2
Horsefly Channel	0.06	Mean Proportion Method - Cyclic 1,2
Horsefly River	0.56	Mean Proportion Method - Cyclic 1,2
Horsefly River-Above Falls		Mean Proportion Method - Cyclic 1,2
Horsefly River-Lower		Mean Proportion Method - Cyclic 1,2
Horsefly River-Upper		Mean Proportion Method - Cyclic 1,2
Little Horsefly River	0.01	Mean Proportion Method - Cyclic 1,2
McKinley Creek	0.07	Mean Proportion Method - Cyclic 1,2
McKinley Creek-Lower		Mean Proportion Method - Cyclic 1,2
McKinley Creek-Upper		Mean Proportion Method - Cyclic 1,2
Mitchell River	0.27	Mean Proportion Method - Cyclic 1,2
Penfold Creek	0.01	Mean Proportion Method - Cyclic 1,2
Seton-L (*de novo*)		
Portage Creek	1.00	NA
Shuswap-ES		
McNomee Creek	0.01	NA
Scotch Creek	0.45	Gap filled with zeros
Seymour River	0.54	NA

- ▢ Available data
- ◘ Gap filled (dominant year)
- ■ Gap filled (sub-dom. year)
- ▮ Gap filled (weak year)

APPENDIX 2. Continued (see previous page for description).

1. Pre- and post-1980 time series separated when calculating proportions
2. Streams grouped according to correlation of trends prior to gap filling
3. Dominant and Sub-dominant cycles only

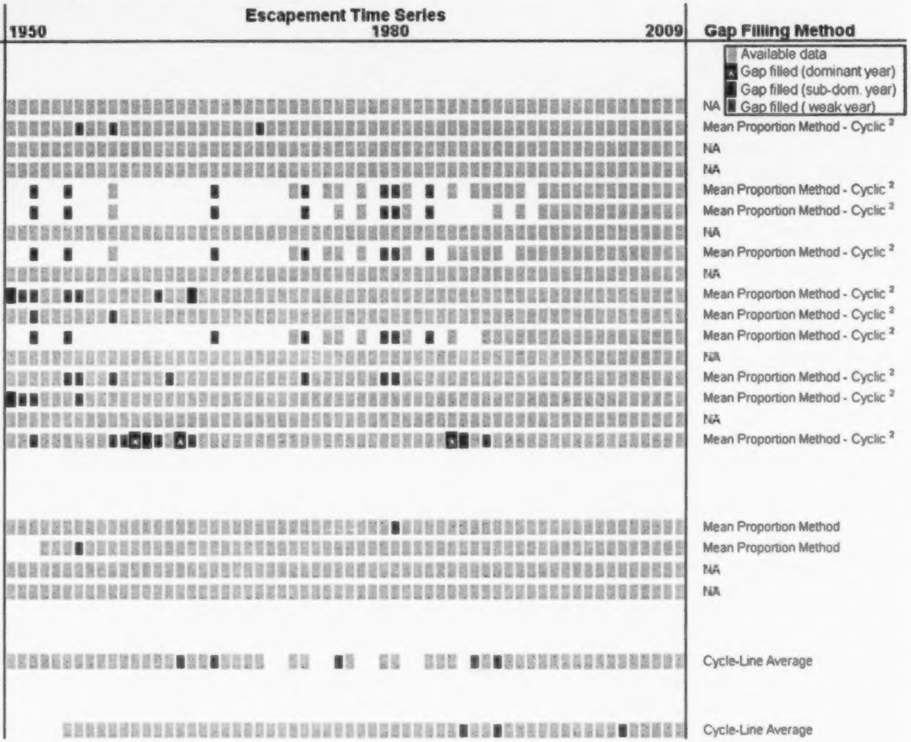


▨ Available data
■ Gap filled (dominant year)
■ Gap filled (sub-dom. year)
■ Gap filled (weak year)

CU Enumeration Site	Avg. Prop.	Gap Filling Method
Shuswap Complex-L		
Adams River	0.70	NA
Anstey River	0.00	Mean Proportion Method - Cyclic ^{3} ^{1}
Eagle River	0.01	Mean Proportion Method - Cyclic ^{3}
Little River	0.10	NA
Lower Shuswap River	0.11	Mean Proportion Method - Cyclic ^{3}
Middle Shuswap River	0.01	NA
Momich River	0.00	Mean Proportion Method - Cyclic ^{3}
Pass Creek	0.00	Mean Proportion Method - Cyclic ^{3}
Scotch Creek	0.00	Mean Proportion Method - Cyclic ^{3}
South Thompson River	0.01	Mean Proportion Method - Cyclic ^{3}
Shuswap Lake	0.01	NA
Adams River-shore	0.00	NA
Anstey Arm	0.01	NA
Anstey River-shore	0.00	NA
Cruikshank Pt West	0.00	NA
Hlina Creek-shore	0.00	NA
Lee Creek-shore	0.00	NA
Main Arm	0.02	NA
Main Arm N	0.00	NA
Main Arm S	0.00	NA
Onyx Creek-shore	0.00	NA
Ross Creek-shore	0.01	NA
Salmon Arm	0.01	NA
Salmon Arm E	0.00	NA
Salmon Arm N	0.00	NA
Salmon Arm S	0.00	NA
Scotch Creek-shore	0.00	NA
Seymour Arm	0.00	NA
Takla-Trembleur-EStu		
5 Mile Creek	0.00	Mean Proportion Method - Cyclic ^{2}
15 Mile Creek	0.00	Mean Proportion Method - Cyclic ^{2}
25 Mile Creek	0.00	NA
Ankwill Creek	0.03	NA
Bivouac Creek	0.02	Mean Proportion Method - Cyclic ^{2}
Blackwater Creek	0.00	Mean Proportion Method - Cyclic ^{2}
Consolidated Creek	0.00	Mean Proportion Method - Cyclic ^{2}
Crow Creek	0.01	Mean Proportion Method - Cyclic ^{2}
Driftwood Creek	0.26	Mean Proportion Method - Cyclic ^{2}
Dust Creek	0.06	NA
Felix Creek	0.06	Mean Proportion Method - Cyclic ^{2}

APPENDIX 2. Continued (see previous page for description).

1. Pre- and post-1980 time series separated when calculating proportions
2. Streams grouped according to correlation of trends prior to gap filling
3. Dominant and Sub-dominant cycles only



CU Enumeration Site	Avg. Prop.	Gap Filling Method
Takla-Trembleur-EStu (contd)		
Forfar Creek	0.09	NA
Forsythe Creek	0.01	Mean Proportion Method - Cyclic [2]
Frypan Creek	0.01	NA
Gluske Creek	0.08	NA
Kastberg Creek	0.00	Mean Proportion Method - Cyclic [2]
Kotesine Creek	0.00	Mean Proportion Method - Cyclic [2]
Kynock Creek	0.16	NA
Lion Creek	0.01	Mean Proportion Method - Cyclic [2]
Narrows Creek	0.04	NA
Paula Creek	0.03	Mean Proportion Method - Cyclic [2]
Point Creek	0.01	Mean Proportion Method - Cyclic [2]
Porter Creek	0.02	Mean Proportion Method - Cyclic [2]
Rosette Creek	0.07	NA
Sakeniche River	0.00	Mean Proportion Method - Cyclic [2]
Sandpoint Creek	0.02	Mean Proportion Method - Cyclic [2]
Shale Creek	0.01	NA
Sinta Creek	0.00	Mean Proportion Method - Cyclic [2]
Takla-Trembleur-Stuart-S		
Kazchek Creek	0.01	Mean Proportion Method
Kuzkwa Creek	0.05	Mean Proportion Method
Middle River	0.35	NA
Tachie River	0.59	NA
Taseko-ES		
Taseko Lake	1.00	Cycle-Line Average
Widgeon-(River-Type)		
Widgeon Creek	1.00	Cycle-Line Average

APPENDIX 3: Historical trends and results of status assessments are illustrated for each assessable CU according to the availability of data. Available figures are organized by CU according to the following structure.

Figure 1: Historical time-series of returns, exploitation, escapement, productivity, and survival plotted for each CU. Figures not available for a CU due to data gaps are noted in individual CU sections. Abundance time-series are not gap-filled in figures.

1a. Total CU returns are separated into total escapement (dark grey-bars), catch (light grey-bars), and en-route loss (red-bars). Exploitation rates are also presented (blue-line).

1b. Total escapement is separated into male (dark grey-bar), female (lighter grey-bar) and female pre-spawn mortality (black-bar) components.

1c. Three standardized (z-score) and smoothed (4 yr running average) indices of productivity time-series: $\log_e$(R/EFS)(light blue triangles-lines), Ricker model residuals (dark blue squares-lines), and Kalman filter a-parameter (dark blue circles-lines) values (the latter index provided by C. Michielsens from the PSC). Large yellow triangles ($\log_e$(R/EFS)) and squares (Ricker residuals) indicate 2005 brood year productivity indices. The Lower Fraser River-River Type CU also has data for the 2006 brood year, shown as large blue triangles ($\log_e$(R/EFS)) and squares (Ricker residuals).

1d. Productivity ($\log_e$ recruits-per-spawner) (red circles-lines) in relation to replacement (e.g. 1 recruit per 1 spawner) (horizontal black line).

1e. Smoothed (4 yr running average) freshwater survival index-**fry**, **fall fry**, or **smolt**-per-EFS (green circles-lines).

1f. Smoothed (4 yr running average) post-fry survival index recruits-per-**smolt**, **fry**, or **fall fry** (blue circles-lines).

Figure 2: Results of rate of change and abundance-based status assessments.

2a. Frequency distribution of the posterior distribution of the linear rate of change of smoothed log-transformed EFS abundances. The posterior distribution (bars) and its median value (black solid line) are plotted in relation to the lower (dashed line) and upper (dotted line) benchmarks.

2b. Change in EFS abundance over the last three generations. The deterministic regression rate of change of smoothed (year labels indicate the last year of the 4-year running average) log-transformed EFS over the past three generations (solid line coloured according to status on this metric: Red, Amber or Green). The lower benchmark rate of decline (25%) is indicated for comparison (black dashed line).

2c. Ratio of the current generational geometric mean EFS abundance to the long-term average geometric mean. Smoothed, log scale historical time-series of EFS (year labels indicate the last year of the 4-year running average) used to calculate the long-term geometric mean (dashed line) is shown. The current generation (hatched box) and the geometric mean of the current generation (solid coloured line) are indicated and coloured according to the status obtained on this metric

2d. Prior (blue line) and posterior (bars) distribution for spawners at maximum recruitment for CUs where stock and recruitment data are available. The median posterior value is indicated with dashed vertical black line. Uniform or lognormal distribution inputs are reported in figure title.

2e. Conservation Unit stock (ETS)-recruitment relationship (model fit: black solid line) with lower (red vertical solid line) and upper (green vertical dashed line) benchmarks indicated.

LIST OF 26 ASSESSIBLE (DATA AVAILABLE) CONSERVATION UNITS WITH FIGURES.

Anderson-Seton-ES 128
Bowron-ES 130
Chilko-ES & Chilko-S 132
Chilliwack-ES 134
Cultus-L 136
Francois-Fraser-S 138
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Harrison (U/S)-L 142
Harrison River (River-Type) 144
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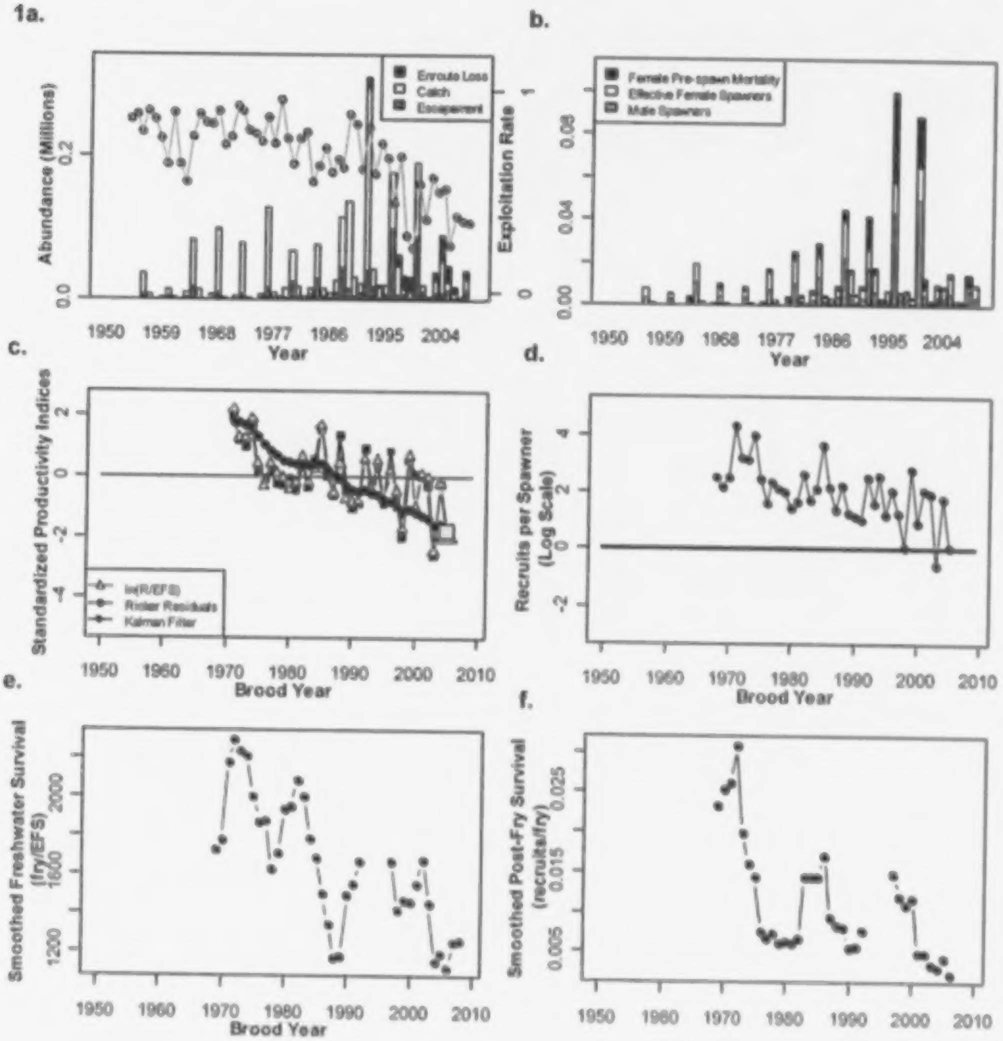
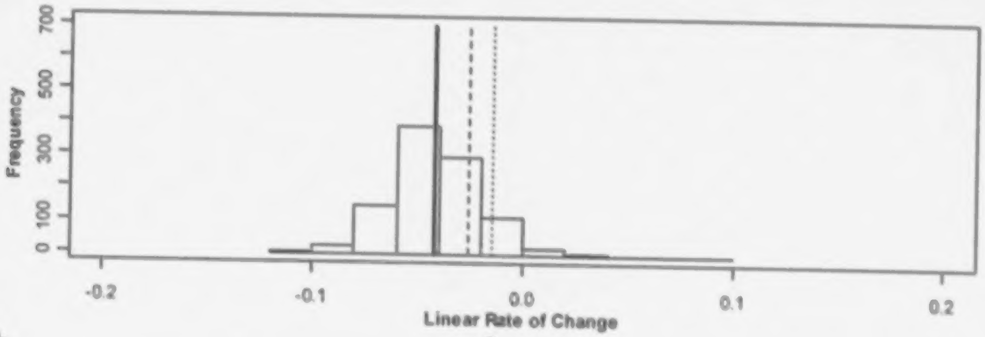
Nadina-Francois-ES 150
Nahatlatch-ES 152
North Barriere-ES (*de novo*) 154
Pitt-ES 156
Quesnel-S 158
Seton-L (*de novo*) 160
Shuswap-ES 162
Shuswap Complex-L 164
Takla-Trembleur-Early Stuart (EStu) 166
Takla-Trembleur-Stuart-S 168
Taseko-ES 170
Widgeon-(River-Type) 172

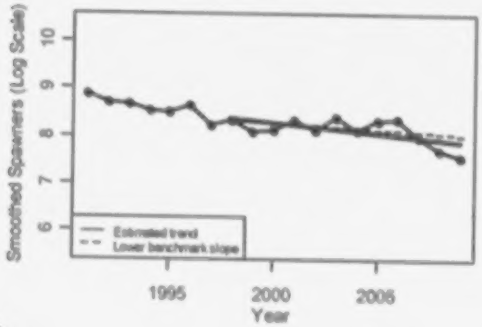
Anderson-Seton-ES



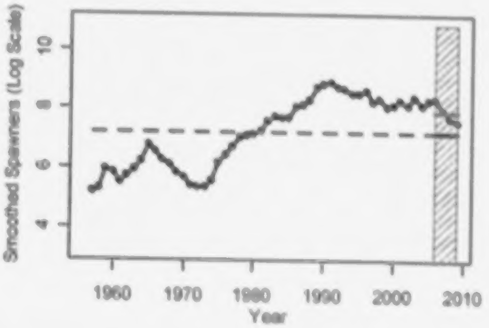
2a.


Frequency
700
500
300
100
0
-0.2
-0.1
0.0
0.1
0.2
Linear Rate of Change

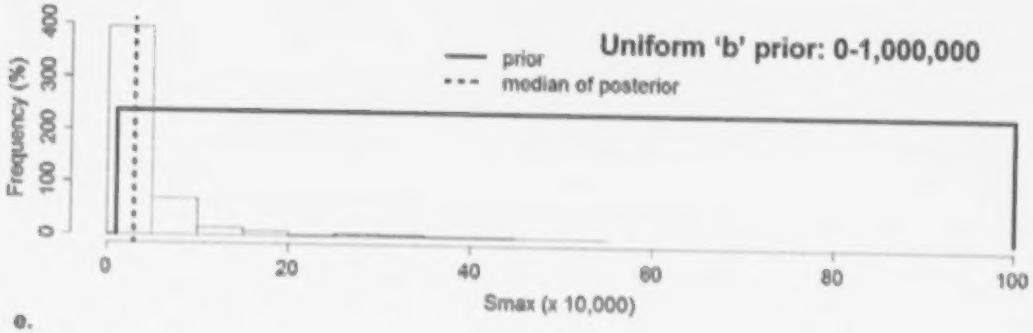
b.


Smoothed Spawners (Log Scale)
10
9
8
7
6
Estimated trend
Lower benchmark slope
1995
2000
2005
Year

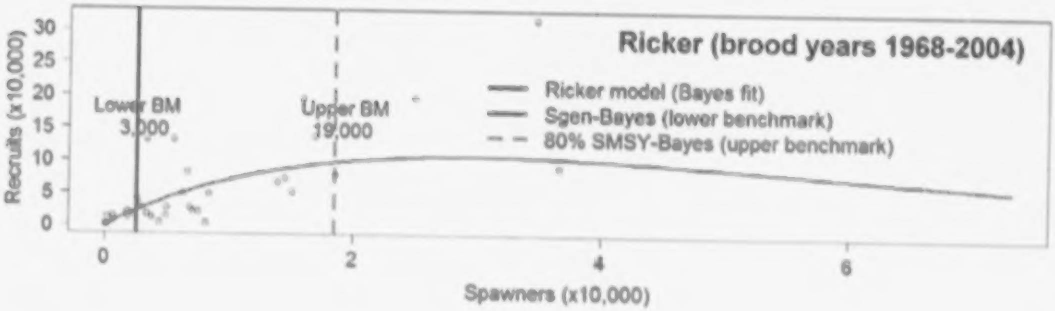
c.


Smoothed Spawners (Log Scale)
10
8
6
4
1960
1970
1980
1990
2000
2010
Year

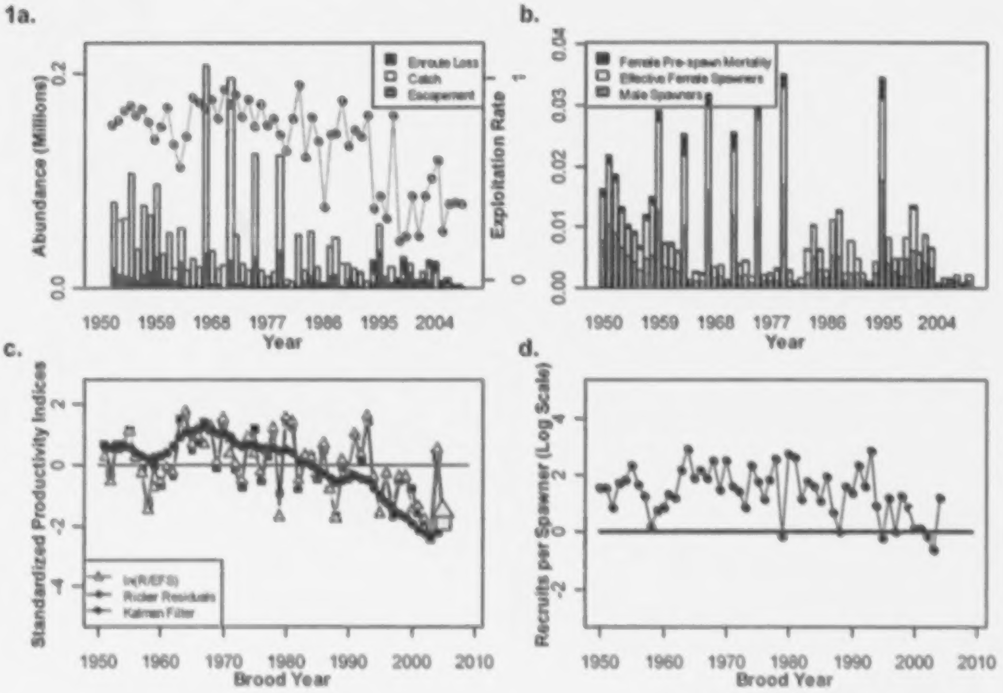
d.


Frequency (%)
400
300
200
100
0
prior
median of posterior
Uniform 'b' prior: 0-1,000,000
0
20
40
60
80
100
Smax (x 10,000)

e.


Recruits (x10,000)
30
25
20
15
10
5
0
Ricker (brood years 1968-2004)
Lower BM
3,000
Upper BM
19,000
Ricker model (Bayes fit)
Sgen-Bayes (lower benchmark)
80% SMSY-Bayes (upper benchmark)
0
2
4
6
Spawners (x10,000)

Bowron-ES

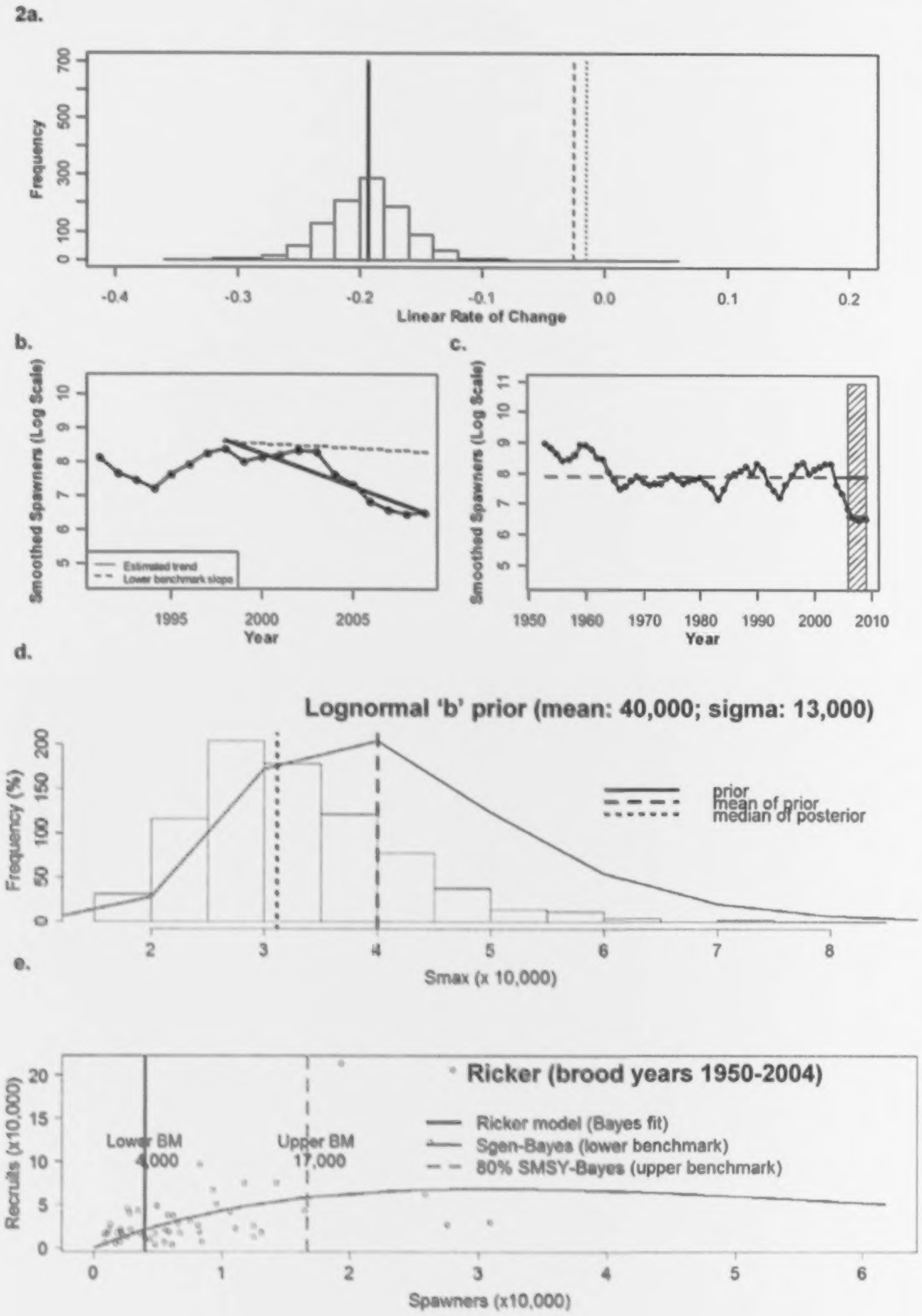


e.

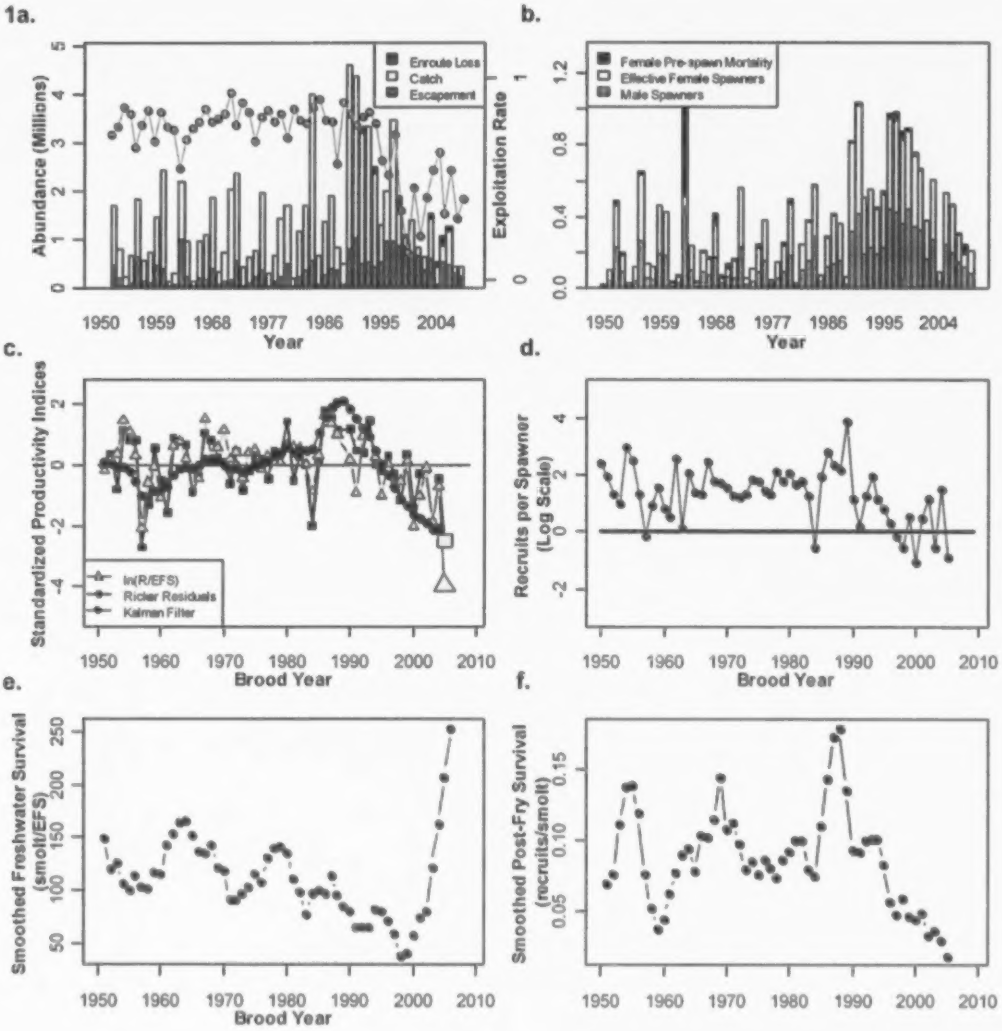
No freshwater survival data available

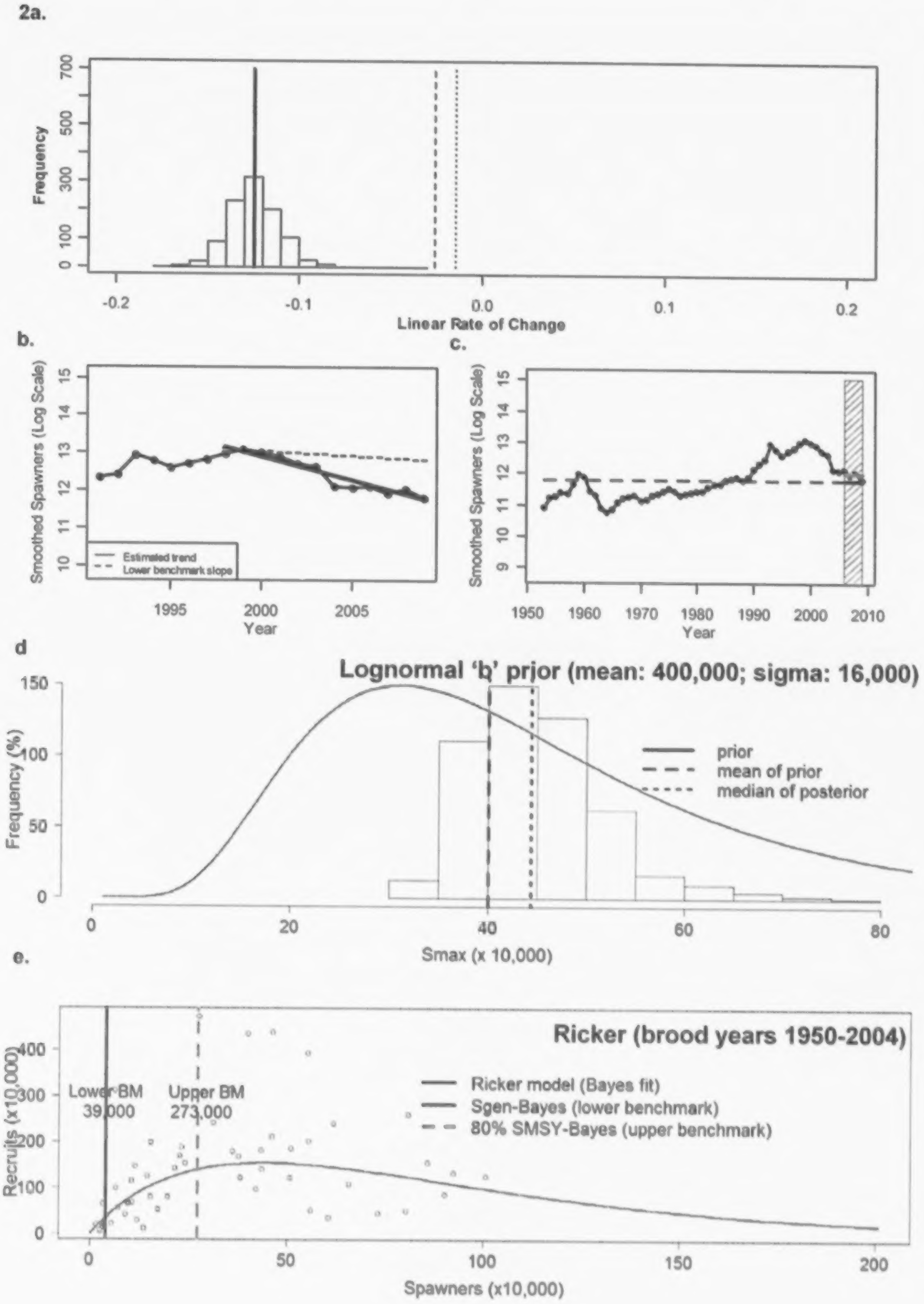
f.

No marine survival data available


2a.
Frequency
0
100
300
500
700
-0.4
-0.3
-0.2
-0.1
0.0
0.1
0.2
Linear Rate of Change
b.
Smoothed Spawners (Log Scale)
5
6
7
8
9
10
Estimated trend
Lower benchmark slope
1995
2000
2005
Year
c.
Smoothed Spawners (Log Scale)
5
6
7
8
9
10
11
1950
1960
1970
1980
1990
2000
2010
Year
d.
Lognormal 'b' prior (mean: 40,000; sigma: 13,000)
Frequency (%)
0
50
100
150
200
prior
mean of prior
median of posterior
2
3
4
5
6
7
8
Smax (x 10,000)
e.
Ricker (brood years 1950-2004)
Recruits (x10,000)
0
5
10
15
20
Lower BM
4,000
Upper BM
17,000
Ricker model (Bayes fit)
Sgen-Bayes (lower benchmark)
80% SMSY-Bayes (upper benchmark)
0
1
2
3
4
5
6
Spawners (x10,000)

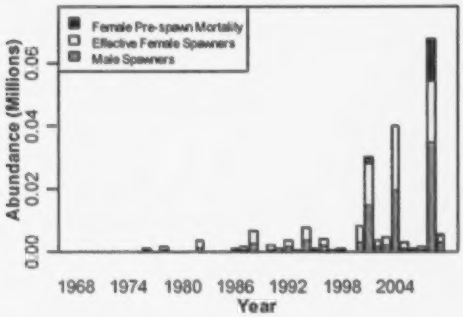
Chilko-ES & Chilko-S



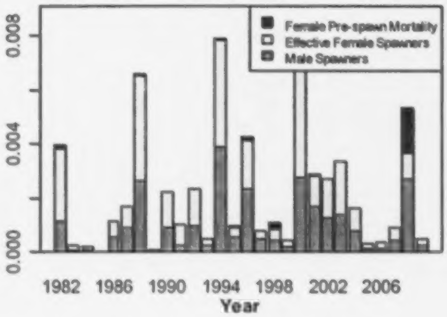

2a.
Frequency
0
100
300
500
700
-0.2
-0.1
0.0
0.1
0.2
Linear Rate of Change
b.
Smoothed Spawners (Log Scale)
10
11
12
13
14
15
Estimated trend
Lower benchmark slope
1995
2000
2005
Year
c.
Smoothed Spawners (Log Scale)
9
10
11
12
13
14
15
1950
1960
1970
1980
1990
2000
2010
Year
d
Lognormal 'b' prior (mean: 400,000; sigma: 16,000)
Frequency (%)
0
50
100
150
prior
mean of prior
median of posterior
0
20
40
60
80
Smax (x 10,000)
e.
Ricker (brood years 1950-2004)
Recruits (x10,000)
0
100
200
300
400
Lower BM
39,000
Upper BM
273,000
Ricker model (Bayes fit)
Sgen-Bayes (lower benchmark)
80% SMSY-Bayes (upper benchmark)
0
50
100
150
200
Spawners (x10,000)

Chilliwack-ES

1a.



1b. (Chilliwack Lake Only).



Only escapement data are available for Chilliwack-ES.
Prior to 2000, only Chilliwack Lake data are available. From 2000 to 2004, Chilliwack River (Dolly Varden) Creek data are also included in the escapement time series.

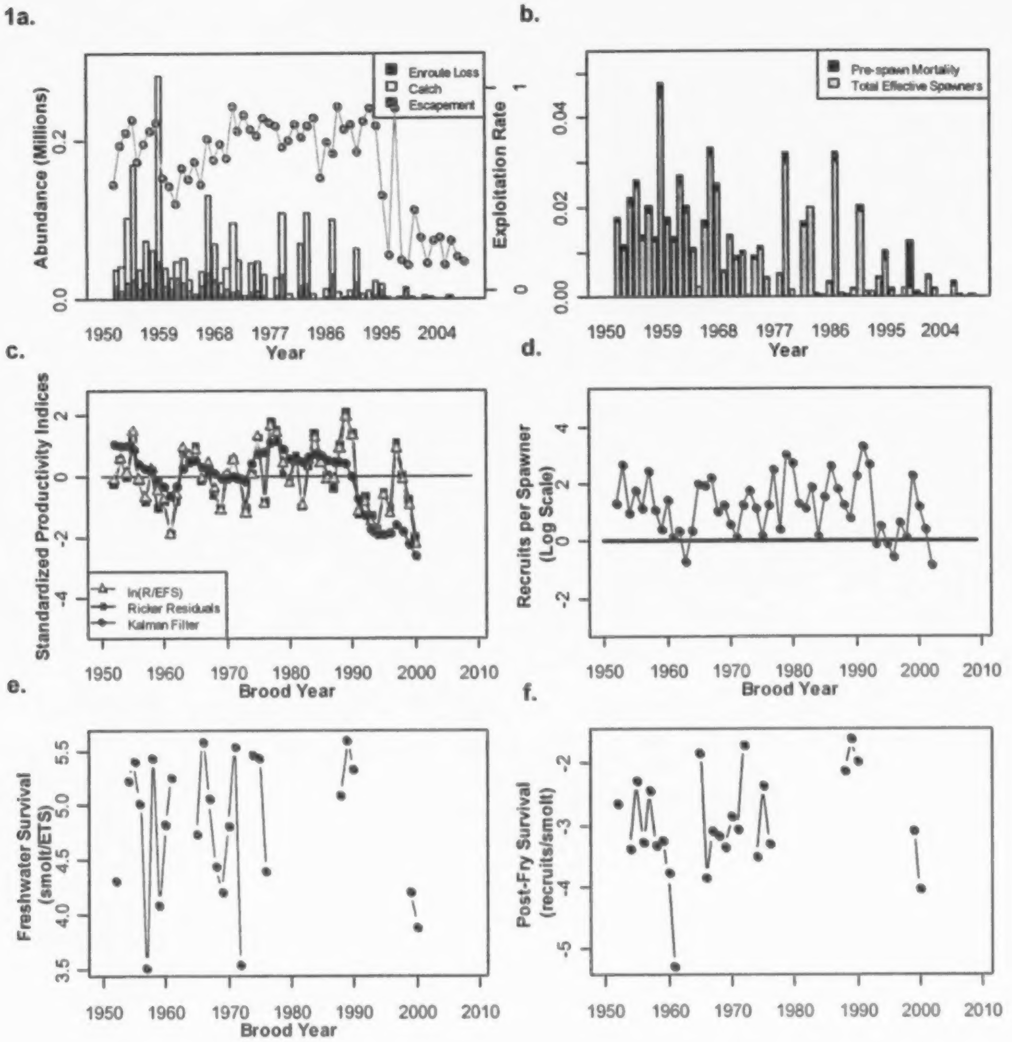
2a.

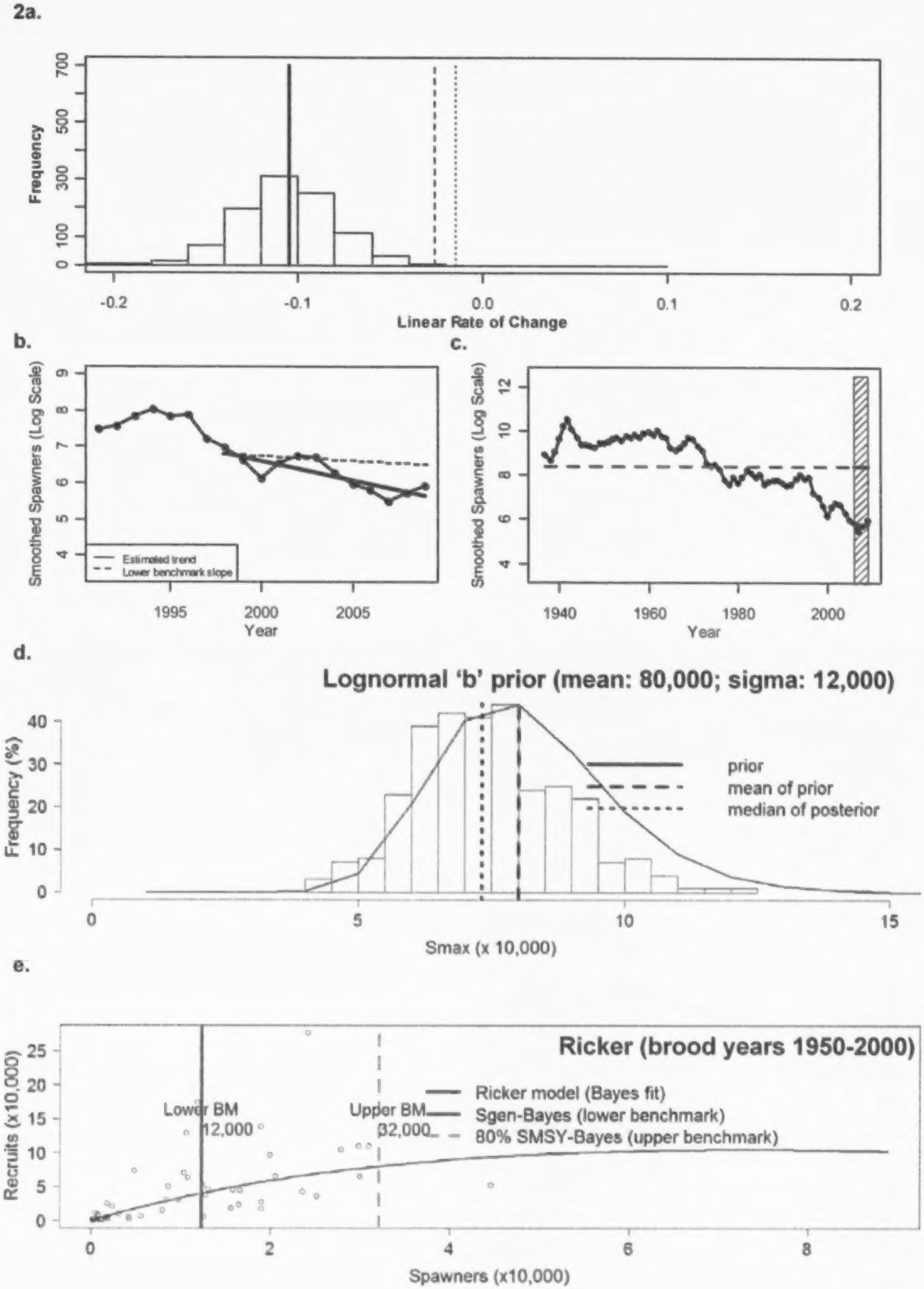
Chilliwack-ES could not be quantitatively assessed in terms of stock status.

b.

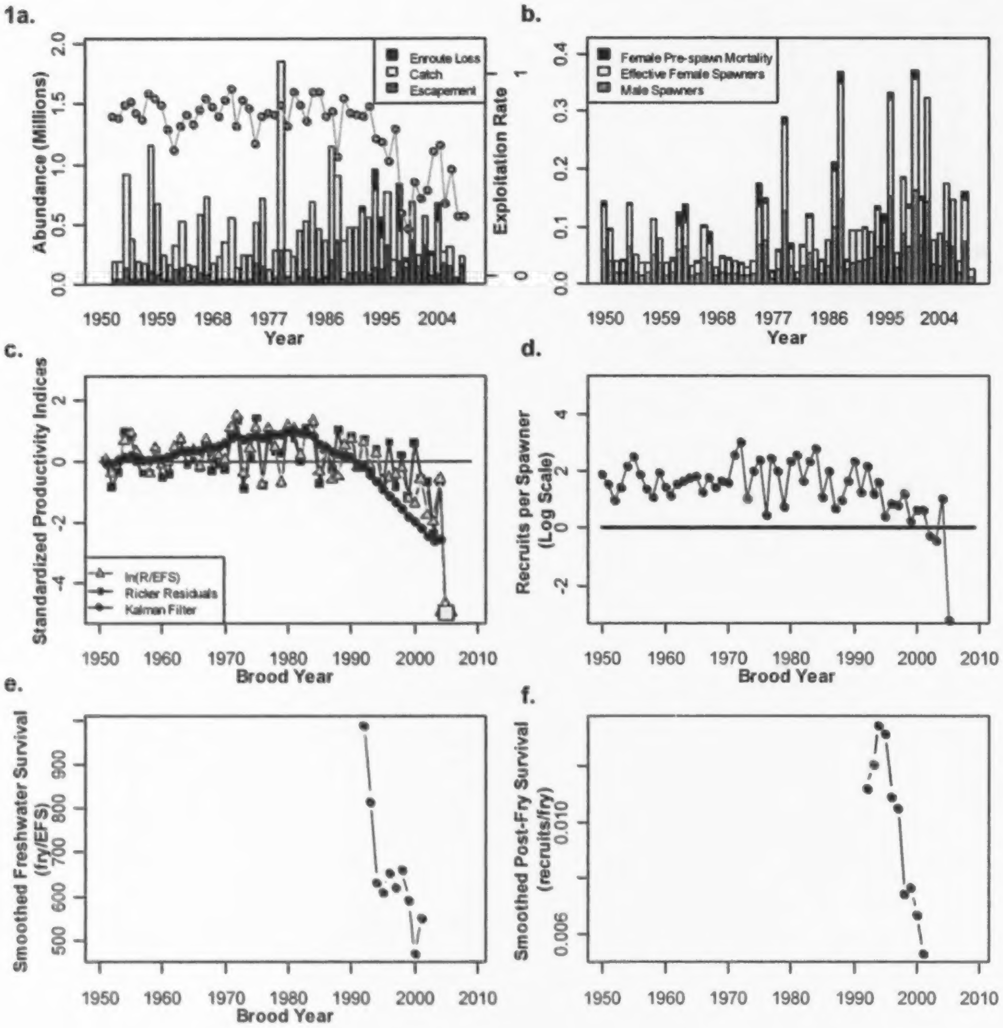
c.

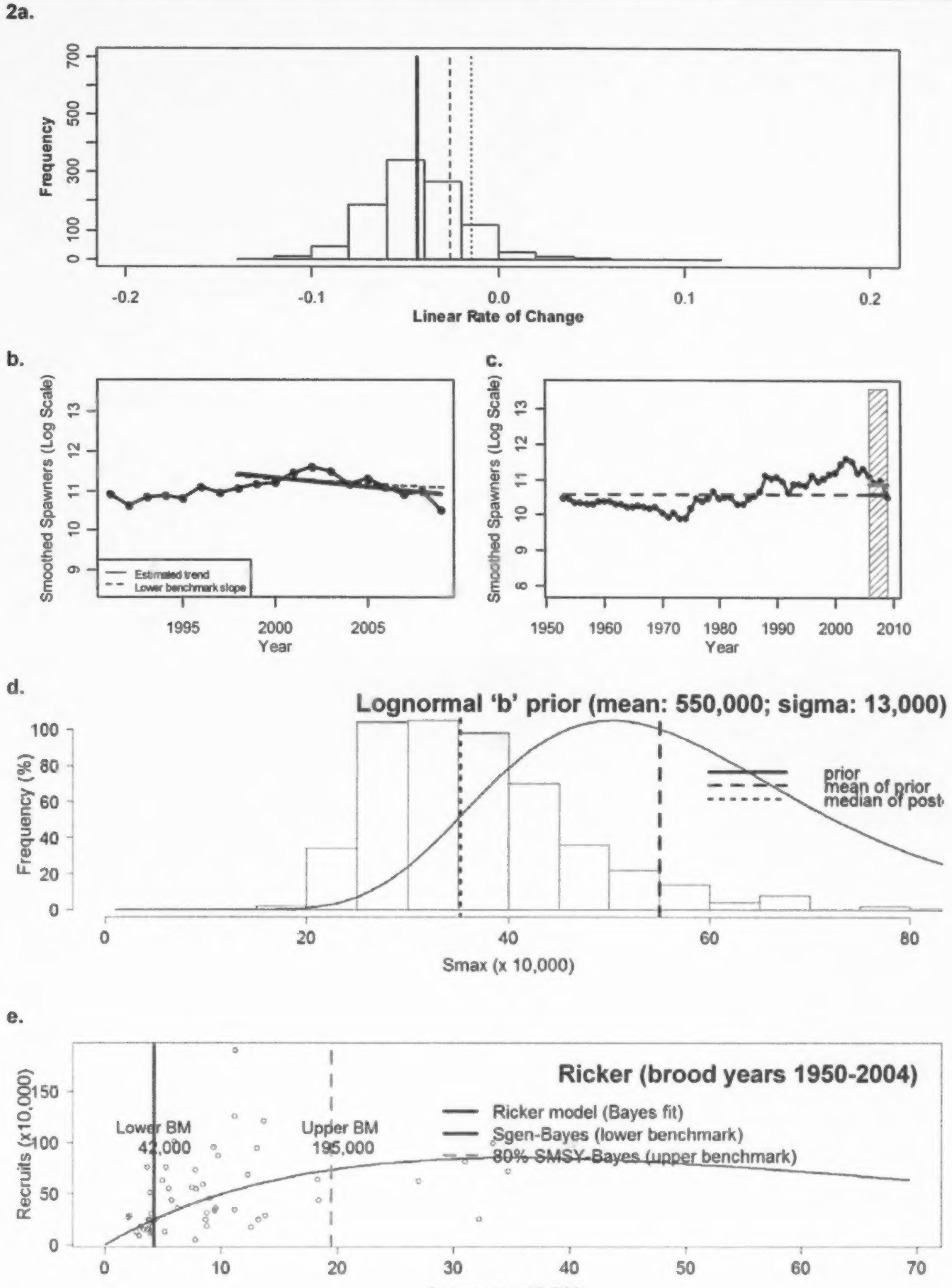
Cultus-L


1a.
Enroute Loss
Catch
Escapement
Abundance (Millions)
0.0
0.2
Exploitation Rate
0
1
1950
1959
1968
1977
1986
1995
2004
Year
b.
Pre-spawn Mortality
Total Effective Spawners
0.00
0.02
0.04
1950
1959
1968
1977
1986
1995
2004
Year
c.
Standardized Productivity Indices
2
0
-2
-4
ln(R/EFS)
Ricker Residuals
Kalman Filter
1950
1960
1970
1980
1990
2000
2010
Brood Year
d.
Recruits per Spawner (Log Scale)
4
2
0
-2
1950
1960
1970
1980
1990
2000
2010
Brood Year
e.
Freshwater Survival (smolt/ETS)
5.5
5.0
4.5
4.0
3.5
1950
1960
1970
1980
1990
2000
2010
Brood Year
f.
Post-Fry Survival (recruits/smolt)
-2
-3
-4
-5
1950
1960
1970
1980
1990
2000
2010


2a.
Frequency
0
100
300
500
700
-0.2
-0.1
0.0
0.1
0.2
Linear Rate of Change
b.
Smoothed Spawners (Log Scale)
4
5
6
7
8
9
Estimated trend
Lower benchmark slope
1995
2000
2005
Year
c.
Smoothed Spawners (Log Scale)
4
6
8
10
12
1940
1960
1980
2000
Year
d.
Lognormal 'b' prior (mean: 80,000; sigma: 12,000)
Frequency (%)
0
10
20
30
40
prior
mean of prior
median of posterior
0
5
10
15
Smax (x 10,000)
e.
Ricker (brood years 1950-2000)
Recruits (x10,000)
0
5
10
15
20
25
Lower BM
12,000
Upper BM
32,000
Ricker model (Bayes fit)
Sgen-Bayes (lower benchmark)
80% SMSY-Bayes (upper benchmark)
0
2
4
6
8
Spawners (x10,000)

Francois-Fraser-S



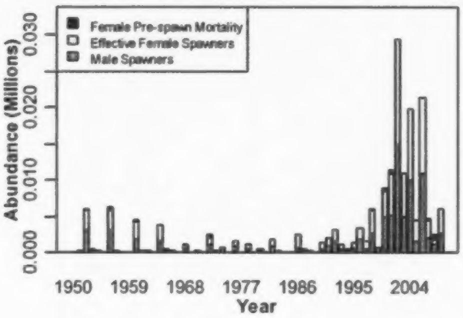

2a.
Frequency
0
100
300
500
700
-0.2
-0.1
0.0
0.1
0.2
Linear Rate of Change
b.
Smoothed Spawners (Log Scale)
9
10
11
12
13
Estimated trend
Lower benchmark slope
1995
2000
2005
Year
c.
Smoothed Spawners (Log Scale)
8
9
10
11
12
13
1950
1960
1970
1980
1990
2000
2010
Year
d.
Lognormal 'b' prior (mean: 550,000; sigma: 13,000)
Frequency (%)
0
20
40
60
80
100
prior
mean of prior
median of post
0
20
40
60
80
Smax (x 10,000)
e.
Ricker (brood years 1950-2004)
Ricker model (Bayes fit)
Sgen-Bayes (lower benchmark)
80% SMSY-Bayes (upper benchmark)
Lower BM
42,000
Upper BM
195,000
Recruits (x10,000)
0
50
100
150
0
10
20
30
40
50
60
70
Spawners (x10,000)

Harrison (D/S)-L

1a.

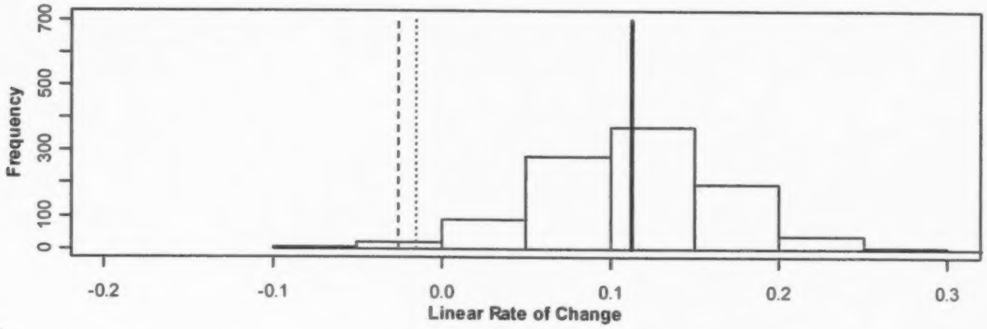
NA

1b.

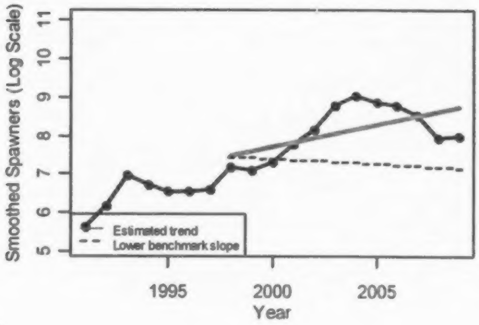


Only escapement data are available for Harrison (D/S)-L.

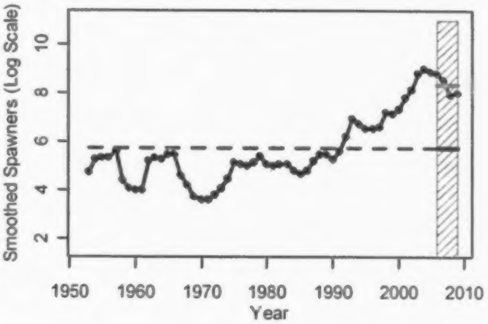
2a.



b.



c.



d.

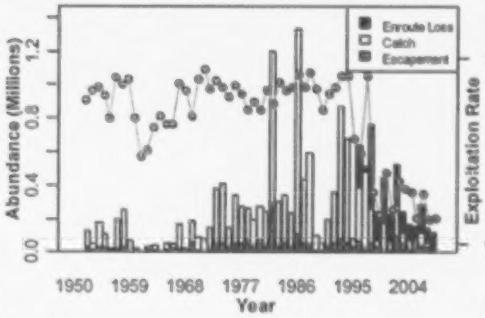
No stock-recruitment data are available to estimate abundance based benchmarks

e.

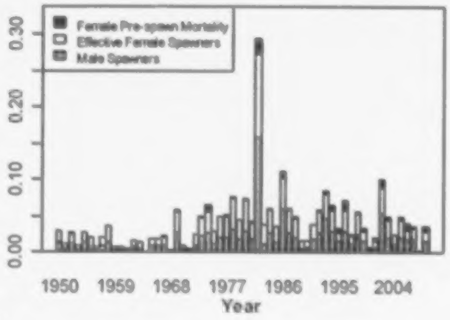
No stock-recruitment data are available to estimate abundance based benchmarks

Harrison (U/S)-L

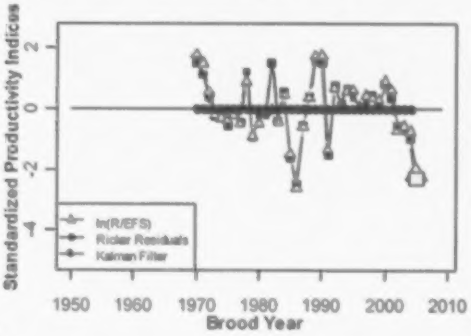
1a.



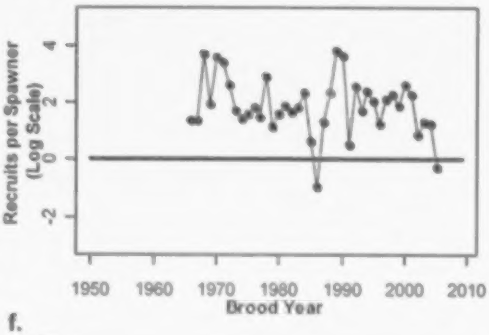
b.



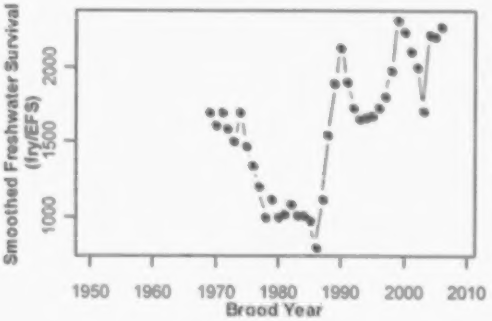
c.



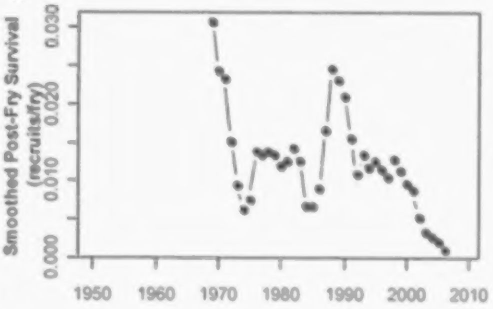
d.

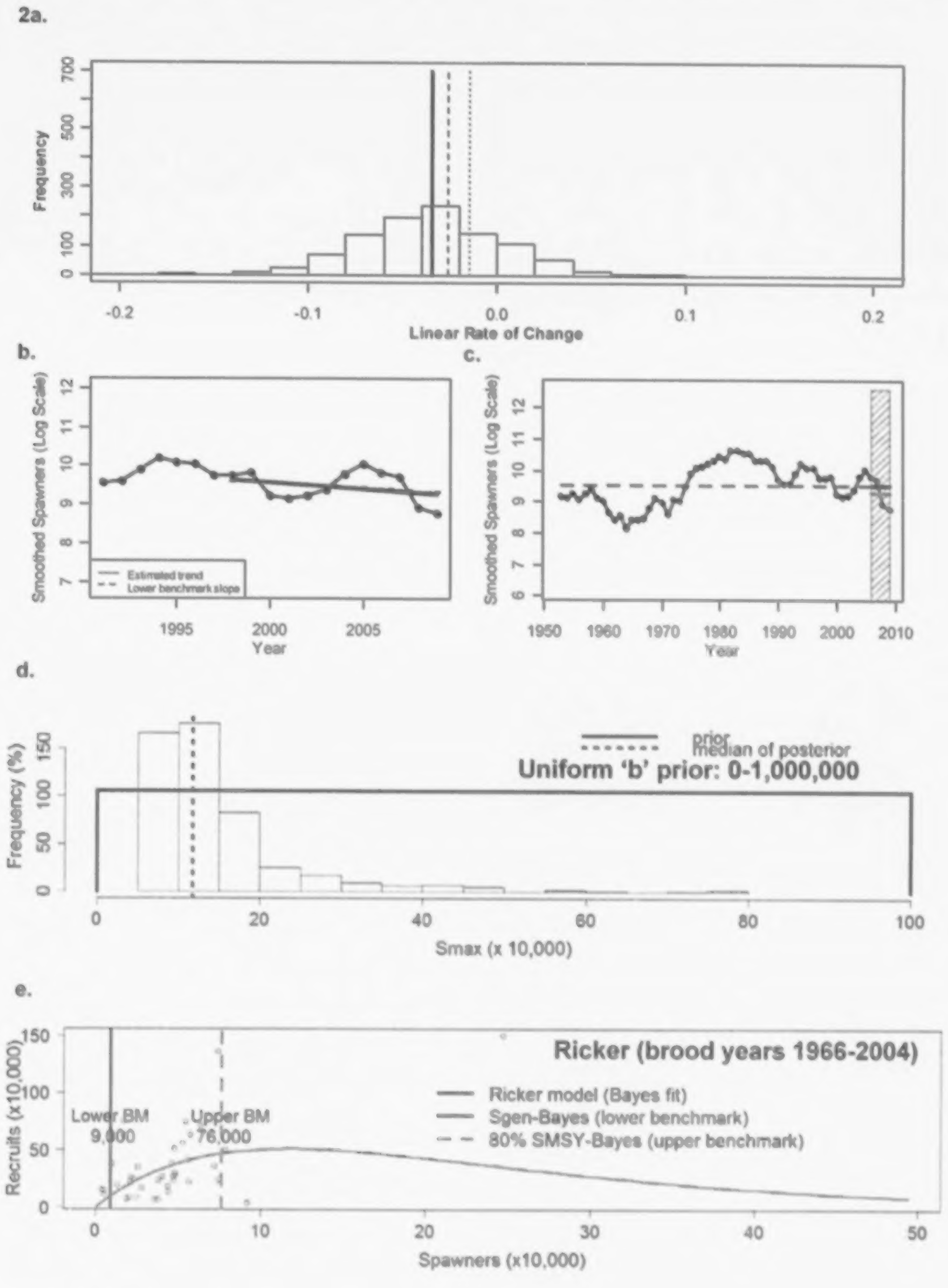


e.



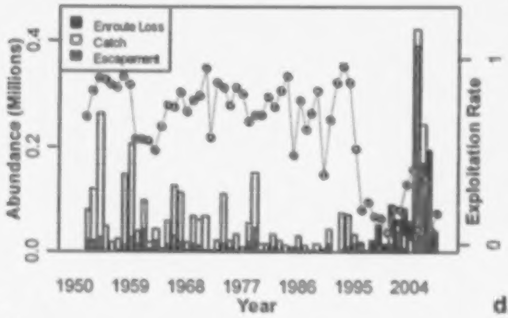
f.



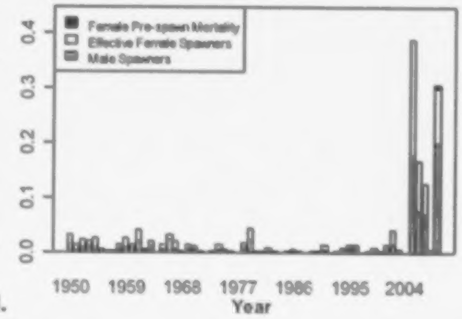

2a.
Frequency
0
100
300
500
700
-0.2
-0.1
0.0
0.1
0.2
Linear Rate of Change
b.
Smoothed Spawners (Log Scale)
7
8
9
10
11
12
Estimated trend
Lower benchmark slope
1995
2000
2005
Year
c.
Smoothed Spawners (Log Scale)
6
7
8
9
10
11
12
1950
1960
1970
1980
1990
2000
2010
Year
d.
prior
median of posterior
Uniform 'b' prior: 0-1,000,000
Frequency (%)
0
50
100
150
0
20
40
60
80
100
Smax (x 10,000)
e.
Ricker (brood years 1966-2004)
Ricker model (Bayes fit)
Sgen-Bayes (lower benchmark)
80% SMSY-Bayes (upper benchmark)
Lower BM
9,000
Upper BM
76,000
Recruits (x10,000)
0
50
100
150
0
10
20
30
40
50
Spawners (x10,000)

Harrison River (River-Type)

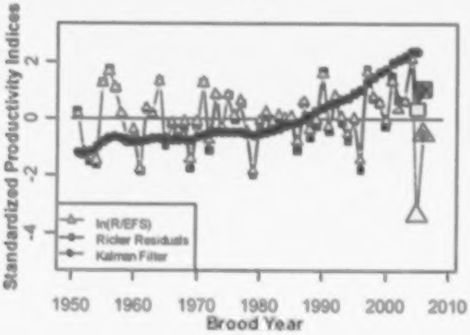
1a.



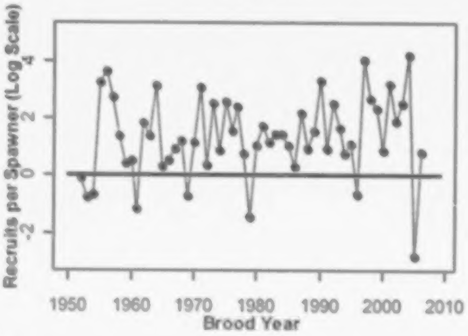
b.



c.



d.

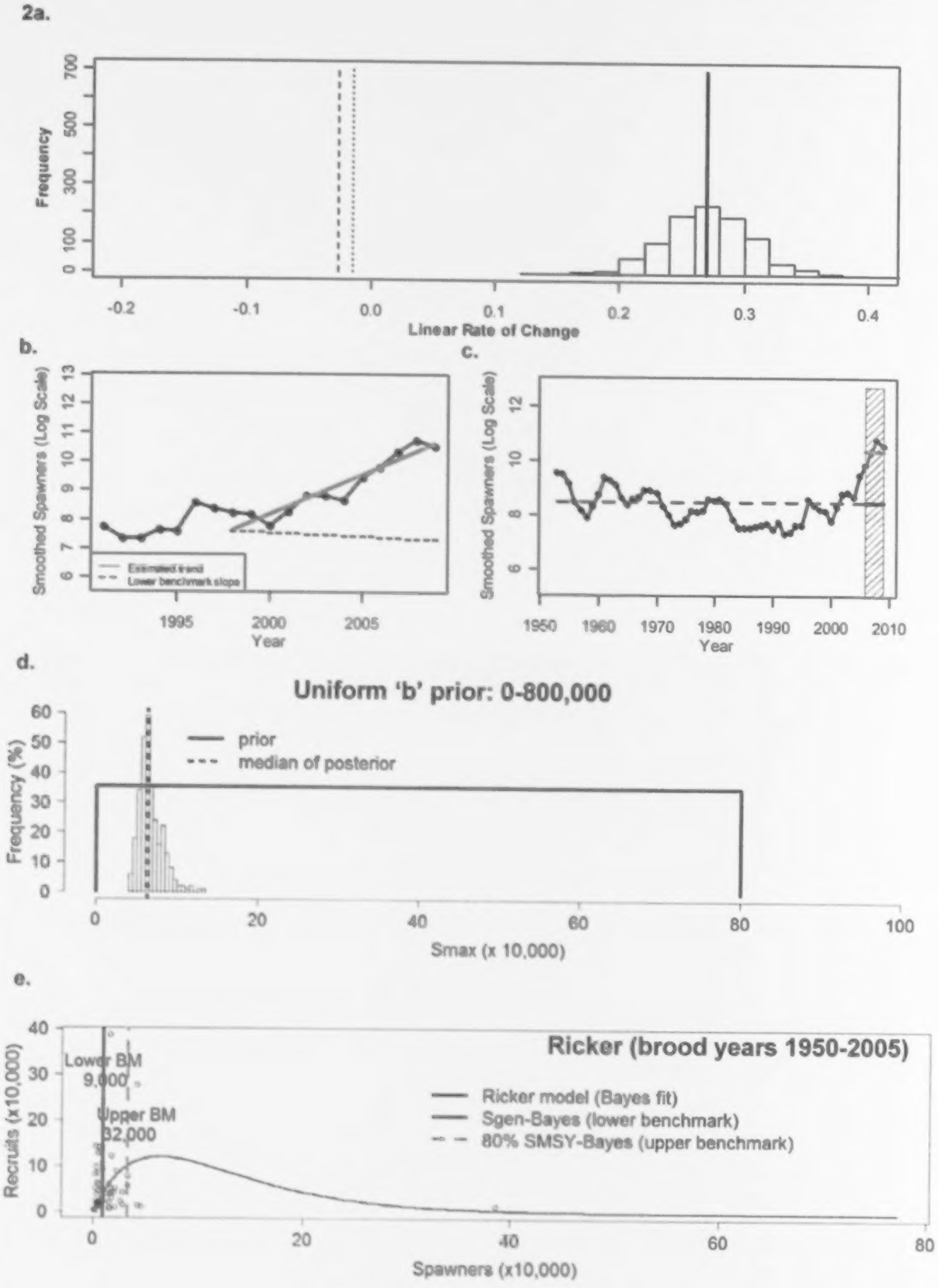


e.

f.

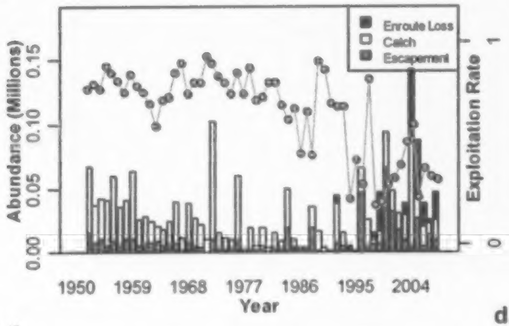
No freshwater survival data are available

No marine survival data are available

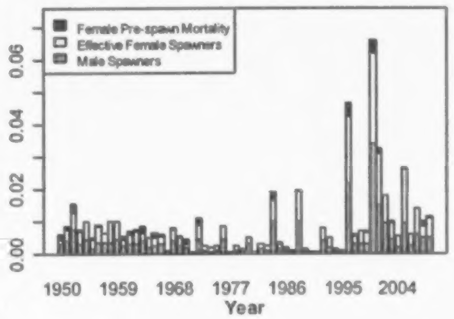

2a.
Frequency
0
100
300
500
700
-0.2
-0.1
0.0
0.1
0.2
0.3
0.4
Linear Rate of Change
b.
Smoothed Spawners (Log Scale)
6
7
8
9
10
11
12
13
Estimated trend
Lower benchmark slope
1995
2000
2005
Year
c.
Smoothed Spawners (Log Scale)
6
8
10
12
1950
1960
1970
1980
1990
2000
2010
Year
d.
Uniform 'b' prior: 0-800,000
Frequency (%)
0
10
20
30
40
50
60
prior
median of posterior
0
20
40
60
80
100
Smax (x 10,000)
e.
Ricker (brood years 1950-2005)
Recruits (x10,000)
0
10
20
30
40
Lower BM
9,000
Upper BM
32,000
Ricker model (Bayes fit)
Sgen-Bayes (lower benchmark)
80% SMSY-Bayes (upper benchmark)
0
20
40
60
80
Spawners (x10,000)

Kamloops-ES

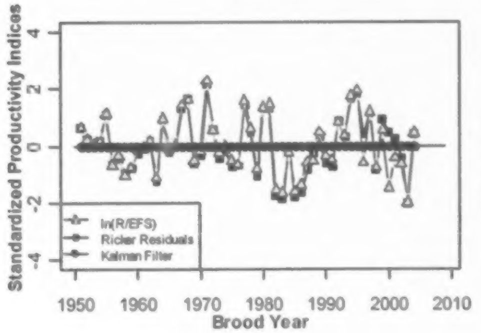
1a.



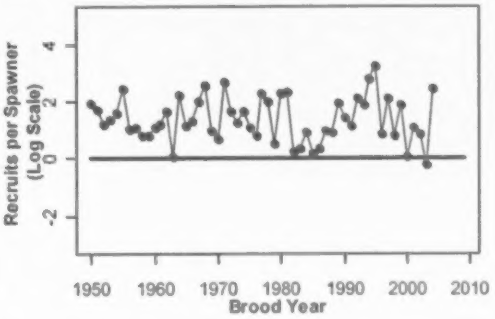
b.



c.



d.



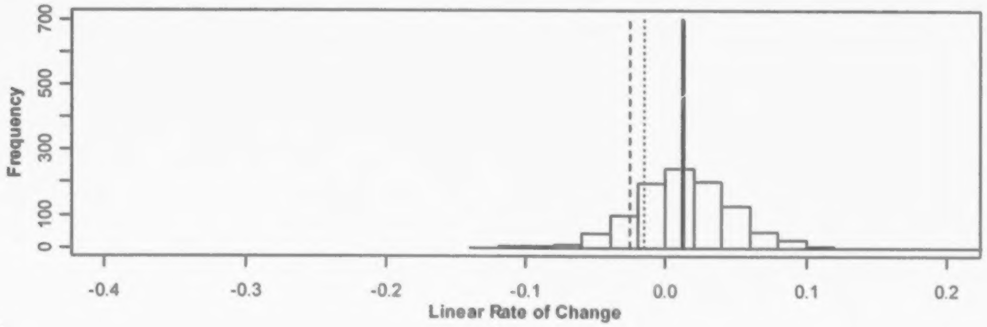
e.

No freshwater survival data are available

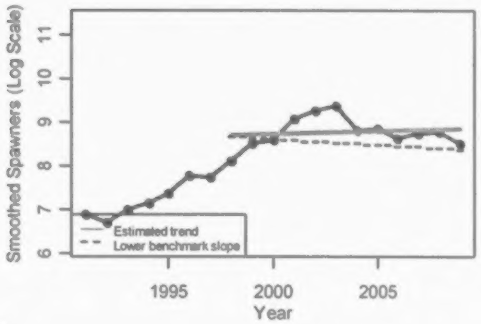
f.

No marine survival data are available

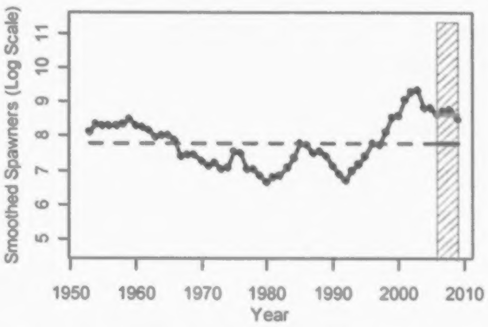
2a.


Frequency
0
100
300
500
700
-0.4
-0.3
-0.2
-0.1
0.0
0.1
0.2
Linear Rate of Change

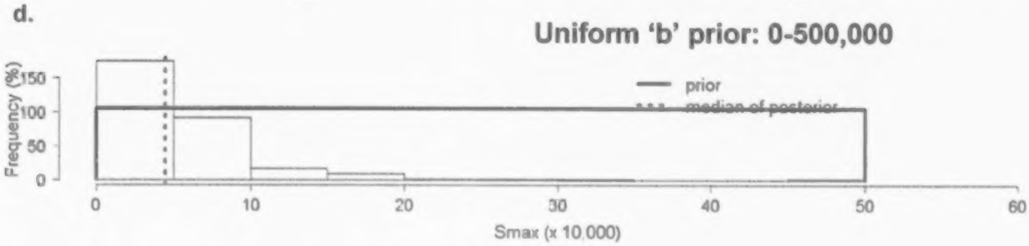
b.


Smoothed Spawners (Log Scale)
6
7
8
9
10
11
Estimated trend
Lower benchmark slope
1995
2000
2005
Year

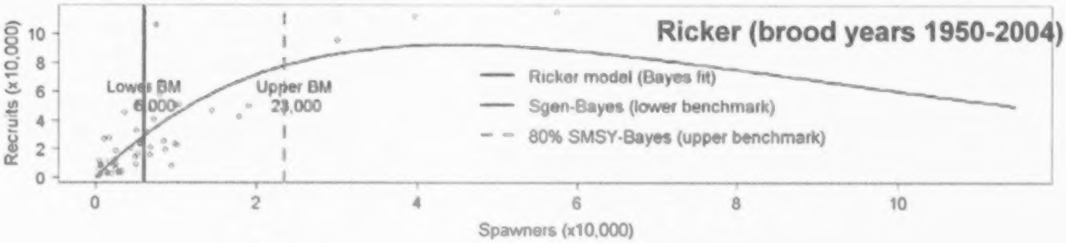
c.


Smoothed Spawners (Log Scale)
5
6
7
8
9
10
11
1950
1960
1970
1980
1990
2000
2010
Year

d.

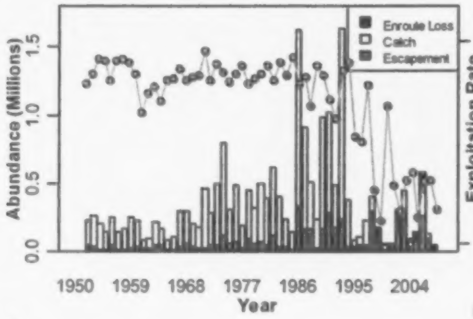

Uniform 'b' prior: 0-500,000
Frequency (%)
0
50
100
150
prior
median of posterior
0
10
20
30
40
50
60
Smax (x 10,000)

e.

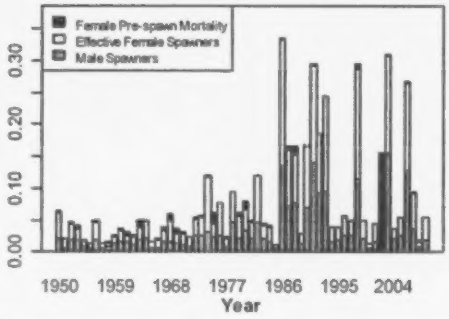

Ricker (brood years 1950-2004)
Recruits (x10,000)
0
2
4
6
8
10
Lower BM
6,000
Upper BM
23,000
Ricker model (Bayes fit)
Sgen-Bayes (lower benchmark)
80% SMSY-Bayes (upper benchmark)
0
2
4
6
8
10
Spawners (x10,000)

Lillooet-Harrison-L

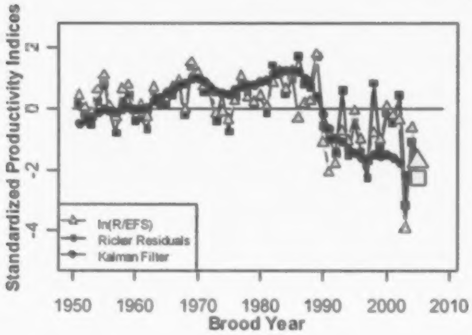
1a.



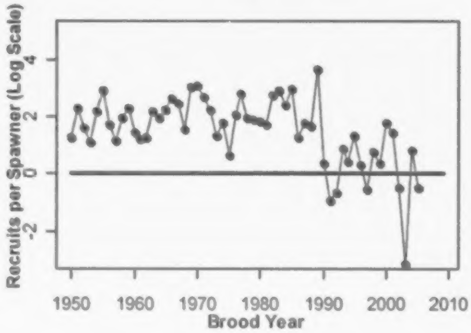
b.



c.



d.

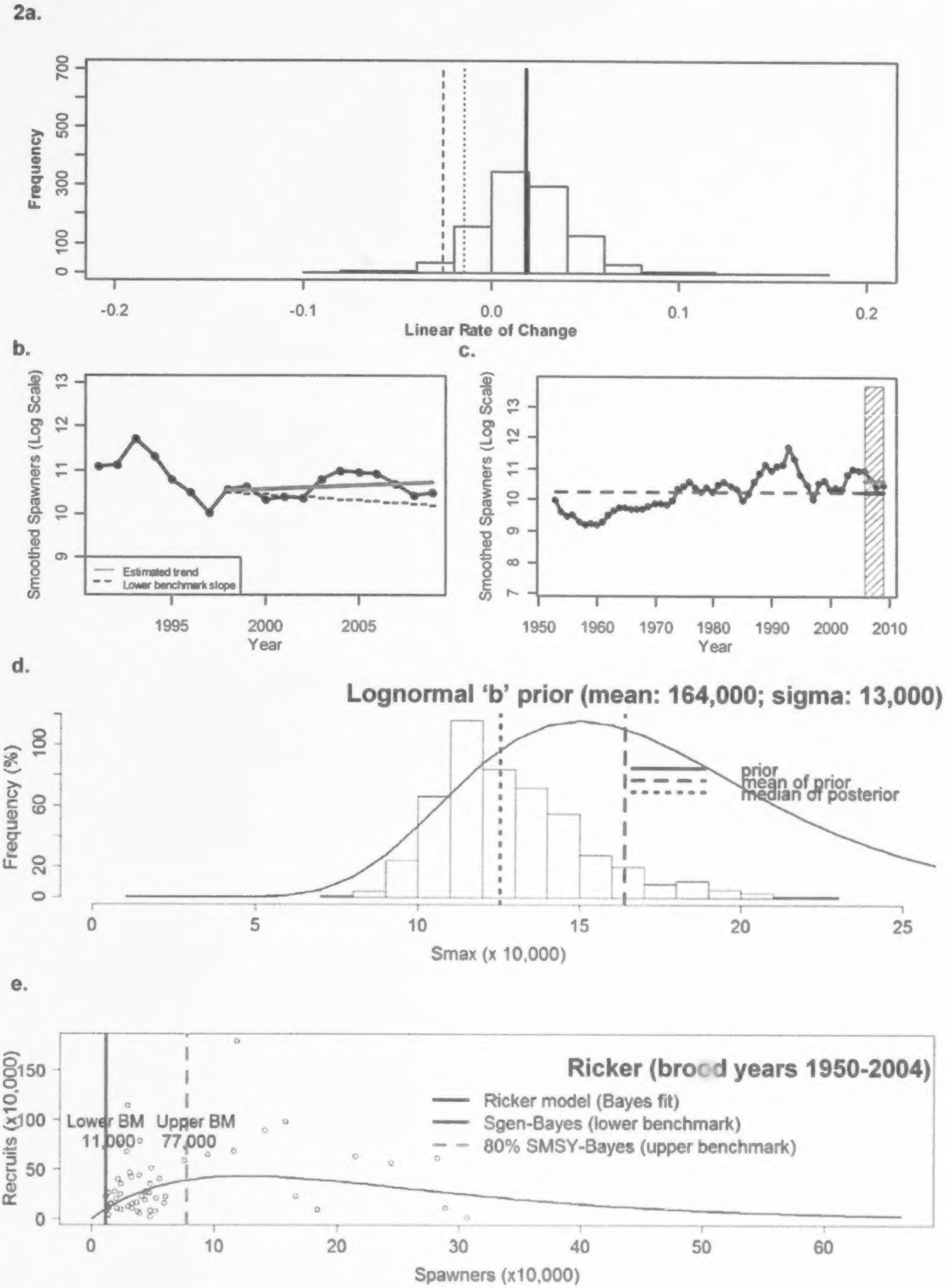


e.

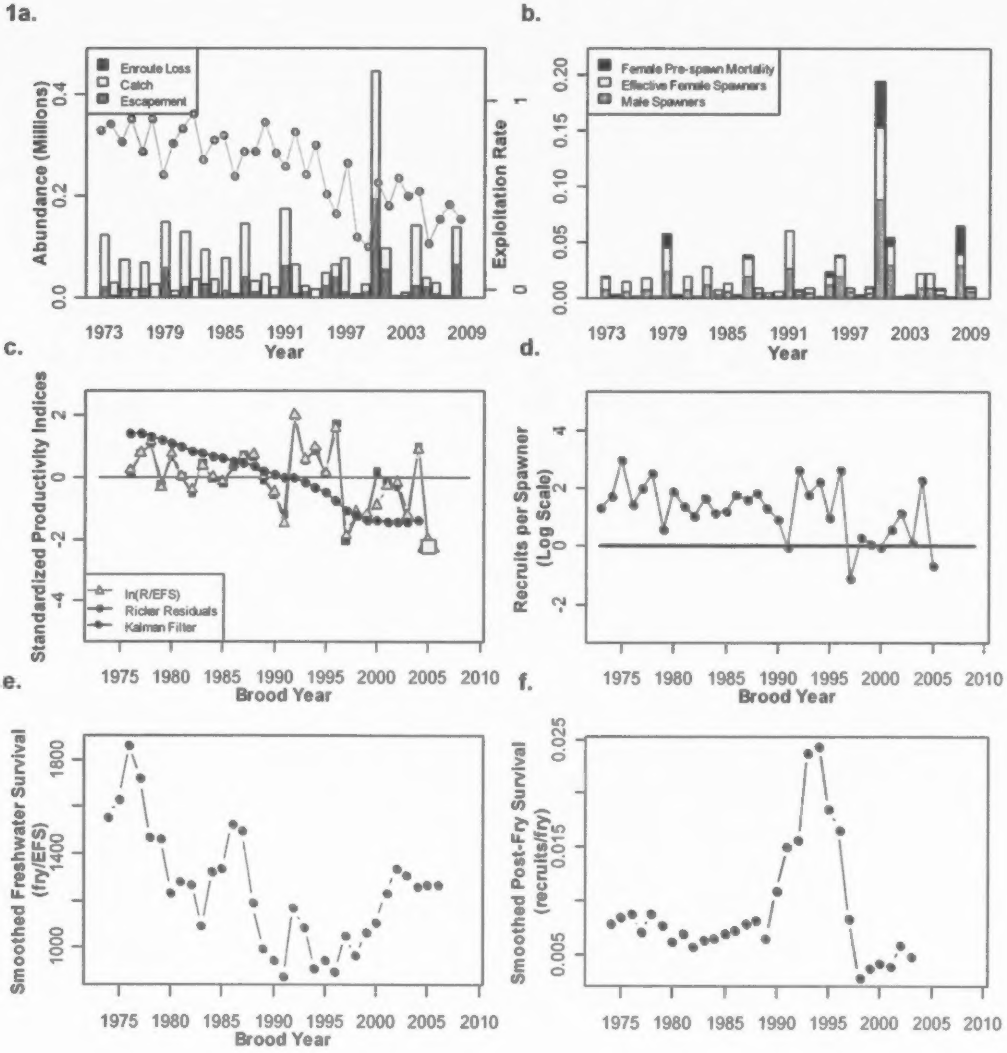
f.

No freshwater survival data are available

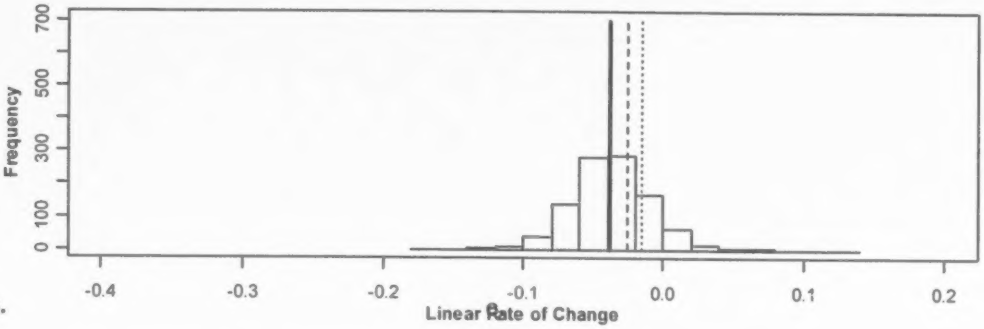
No marine survival data are available


2a.
Frequency
0
100
300
500
700
-0.2
-0.1
0.0
0.1
0.2
Linear Rate of Change
b.
Smoothed Spawners (Log Scale)
9
10
11
12
13
Estimated trend
Lower benchmark slope
1995
2000
2005
Year
c.
Smoothed Spawners (Log Scale)
7
8
9
10
11
12
13
1950
1960
1970
1980
1990
2000
2010
Year
d.
Lognormal 'b' prior (mean: 164,000; sigma: 13,000)
Frequency (%)
0
20
60
100
prior
mean of prior
median of posterior
0
5
10
15
20
25
Smax (x 10,000)
e.
Ricker (brood years 1950-2004)
Ricker model (Bayes fit)
Sgen-Bayes (lower benchmark)
80% SMSY-Bayes (upper benchmark)
Lower BM
11,000
Upper BM
77,000
Recruits (x10,000)
0
50
100
150
0
10
20
30
40
50
60
Spawners (x10,000)

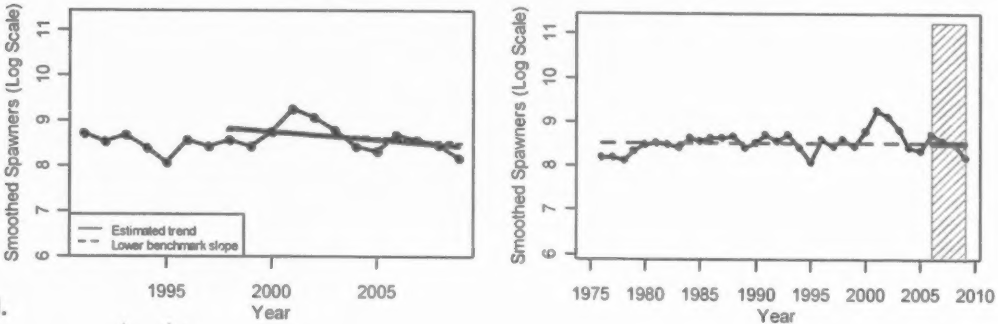
Nadina-Francois-ES



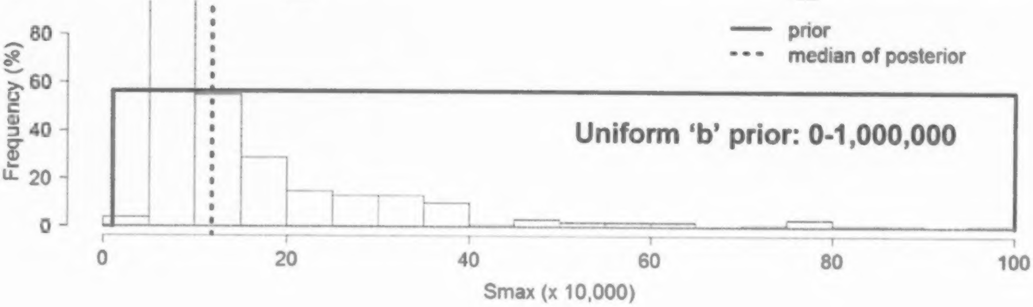
2a.


Frequency
0
100
300
500
700
-0.4
-0.3
-0.2
-0.1
0.0
0.1
0.2
Linear Rate of Change

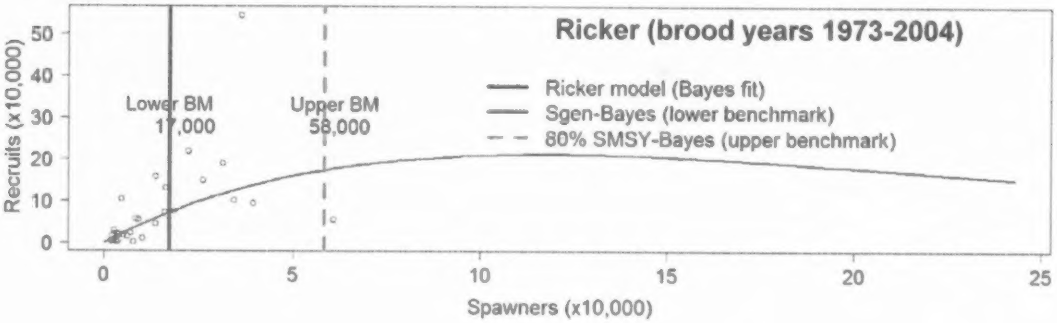
b.


Smoothed Spawners (Log Scale)
6
7
8
9
10
11
Estimated trend
Lower benchmark slope
1995
2000
2005
Year
Smoothed Spawners (Log Scale)
6
7
8
9
10
11
1975
1980
1985
1990
1995
2000
2005
2010
Year

d.


Frequency (%)
0
20
40
60
80
prior
median of posterior
Uniform 'b' prior: 0-1,000,000
0
20
40
60
80
100
Smax (x 10,000)

e.

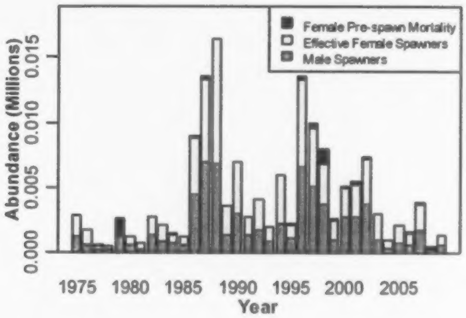

Recruits (x10,000)
0
10
20
30
40
50
Ricker (brood years 1973-2004)
Lower BM
17,000
Upper BM
58,000
Ricker model (Bayes fit)
Sgen-Bayes (lower benchmark)
80% SMSY-Bayes (upper benchmark)
0
5
10
15
20
25
Spawners (x10,000)

Nahatlatch-ES

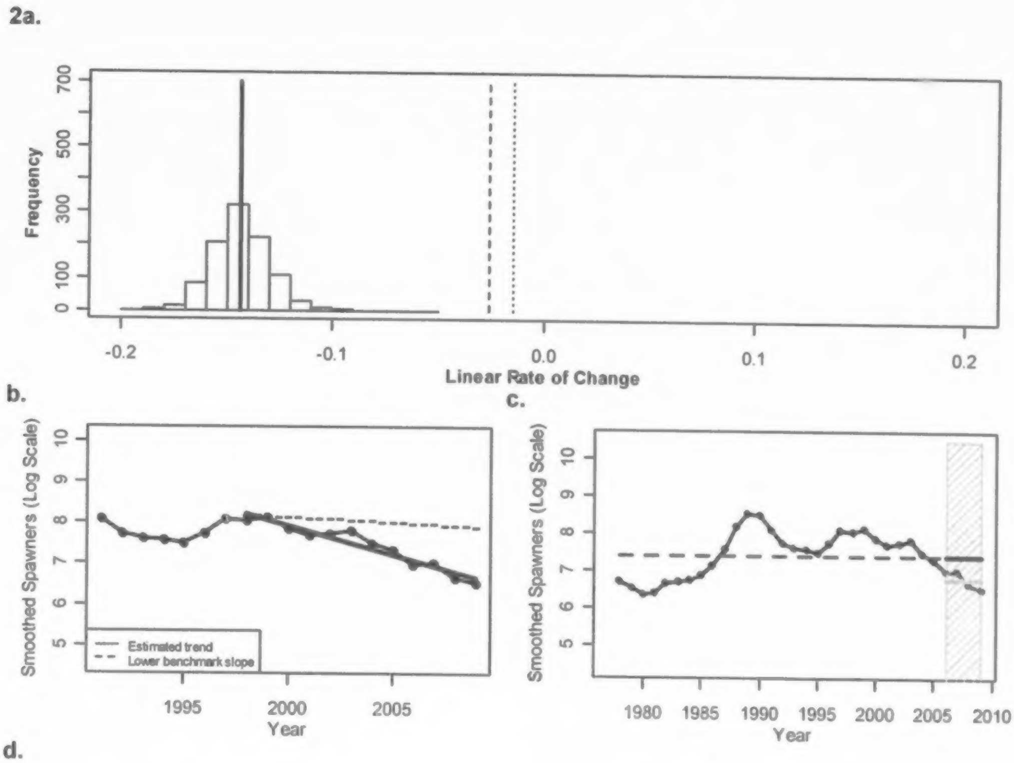
1a.

NA

1b.



Only escapement data are available for Nahatlach-ES.



d.

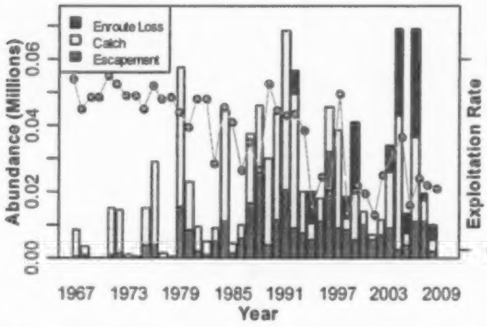
No stock-recruitment data are available to estimate abundance based benchmarks

e.

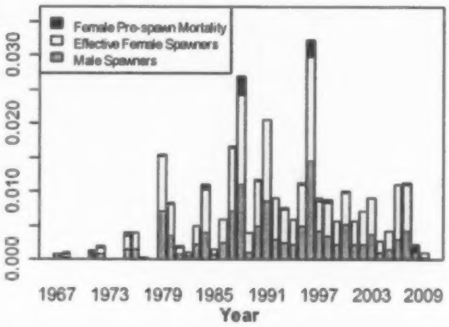
No stock-recruitment data are available to estimate abundance based benchmarks

North Barriere-ES (*de novo*)

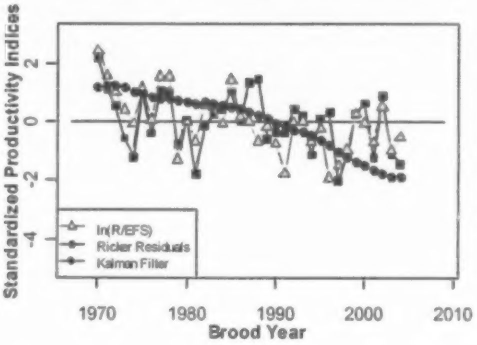
1a.



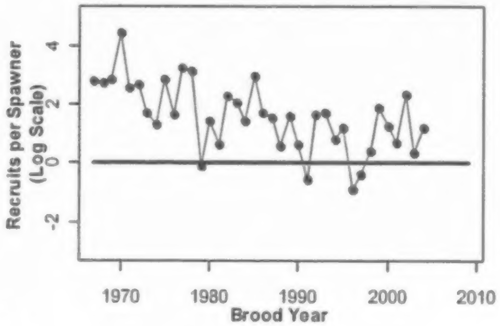
b.



c.



d.

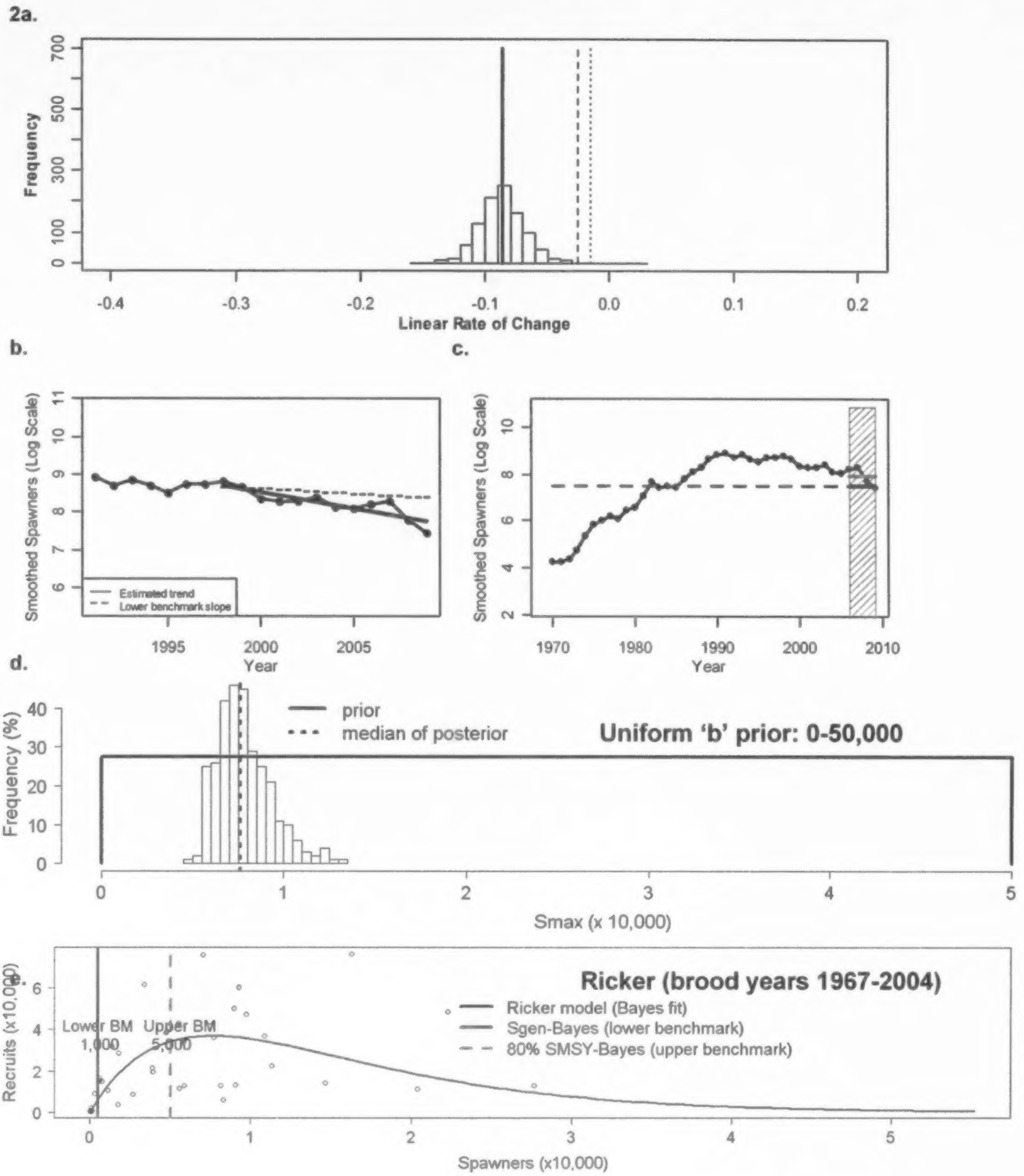


e.

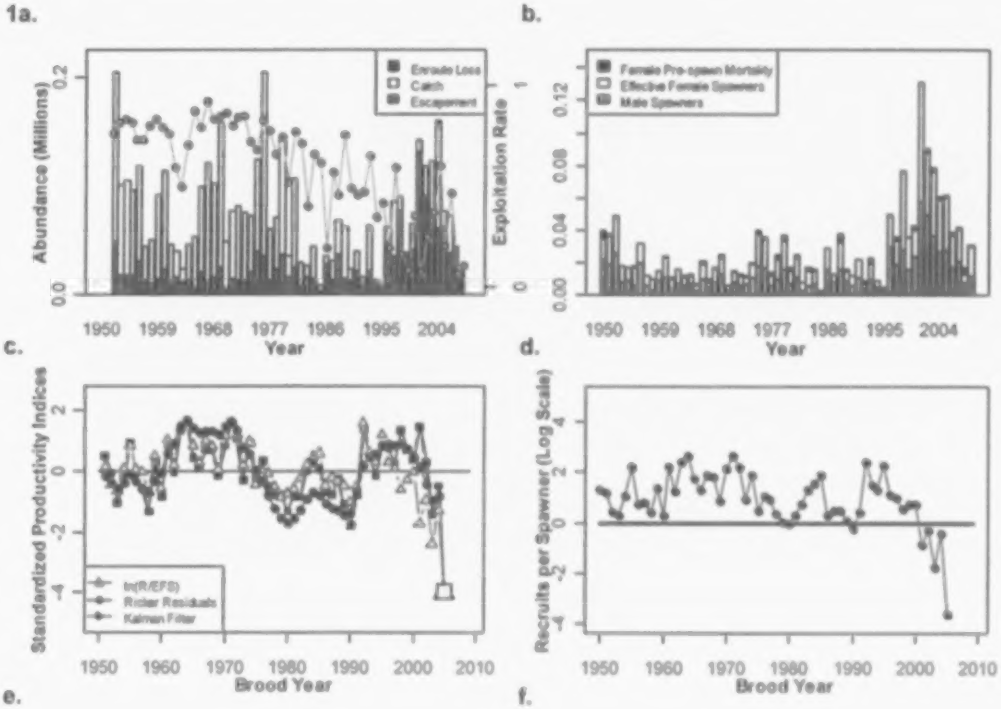
No freshwater survival data are available

f.

No marine survival data are available


2a.
Frequency
0
100
300
500
700
-0.4
-0.3
-0.2
-0.1
0.0
0.1
0.2
Linear Rate of Change
b.
Smoothed Spawners (Log Scale)
6
7
8
9
10
11
Estimated trend
Lower benchmark slope
1995
2000
2005
Year
c.
Smoothed Spawners (Log Scale)
2
4
6
8
10
1970
1980
1990
2000
2010
Year
d.
Frequency (%)
0
10
20
30
40
prior
median of posterior
Uniform 'b' prior: 0-50,000
0
1
2
3
4
5
Smax (x 10,000)
e.
Recruits (x10,000)
0
2
4
6
Lower BM
1,000
Upper BM
5,000
Ricker (brood years 1967-2004)
Ricker model (Bayes fit)
Sgen-Bayes (lower benchmark)
80% SMSY-Bayes (upper benchmark)
0
1
2
3
4
5
Spawners (x10,000)

Pitt-ES



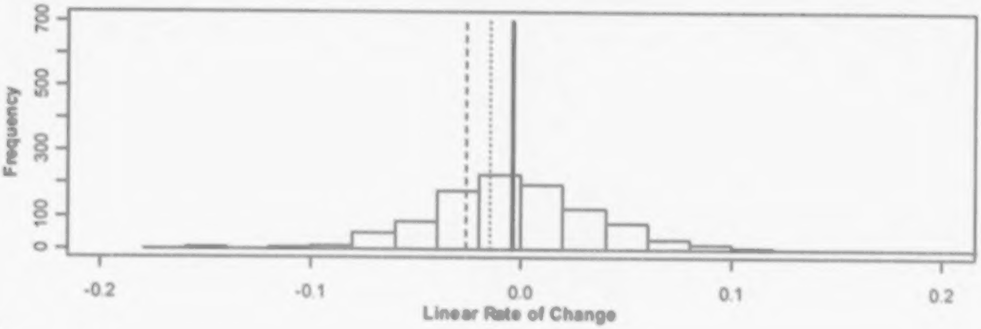
e.

f.

No freshwater survival data are available

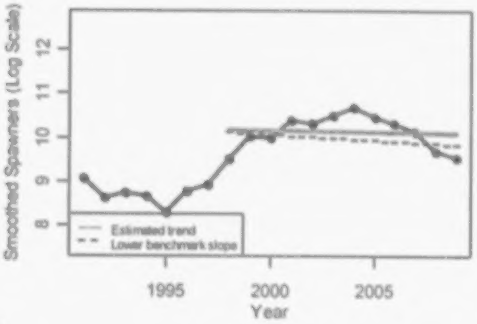
No marine survival data are available

2a.



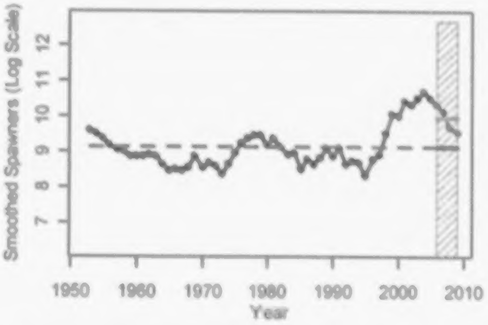
Frequency
0
100
300
500
700
-0.2
-0.1
0.0
0.1
0.2
Linear Rate of Change

b.



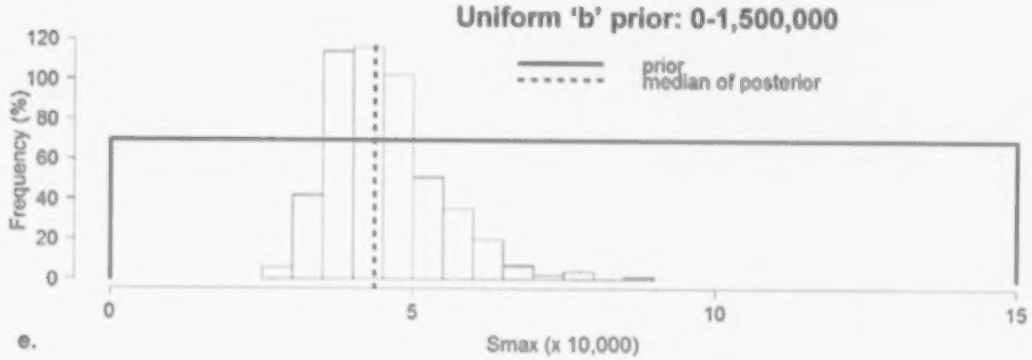
Smoothed Spawners (Log Scale)
8
9
10
11
12
1995
2000
2005
Year
Estimated trend
Lower benchmark slope

c.



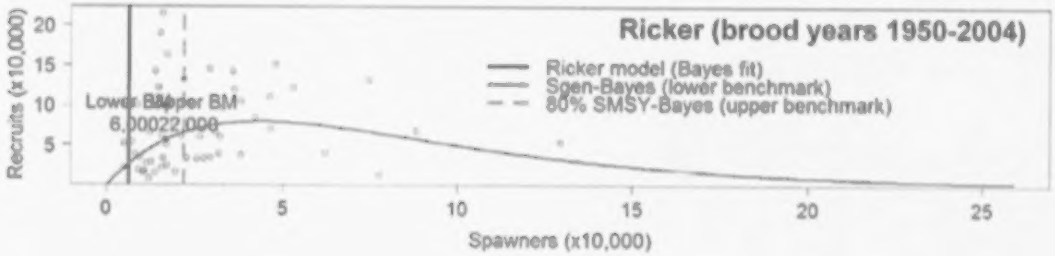
Smoothed Spawners (Log Scale)
7
8
9
10
11
12
1950
1960
1970
1980
1990
2000
2010
Year

d.



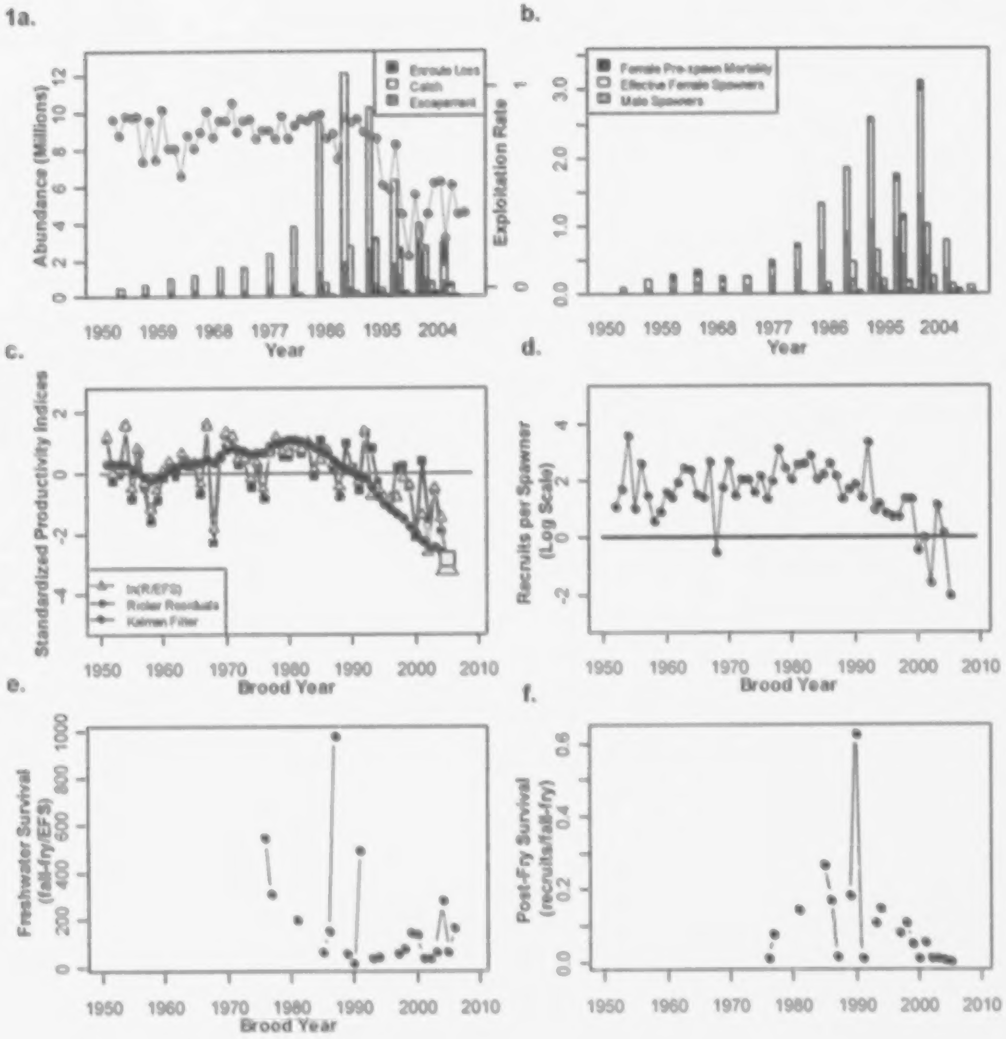
Uniform 'b' prior: 0-1,500,000
prior
median of posterior
Frequency (%)
0
20
40
60
80
100
120
0
5
10
15
Smax (x 10,000)

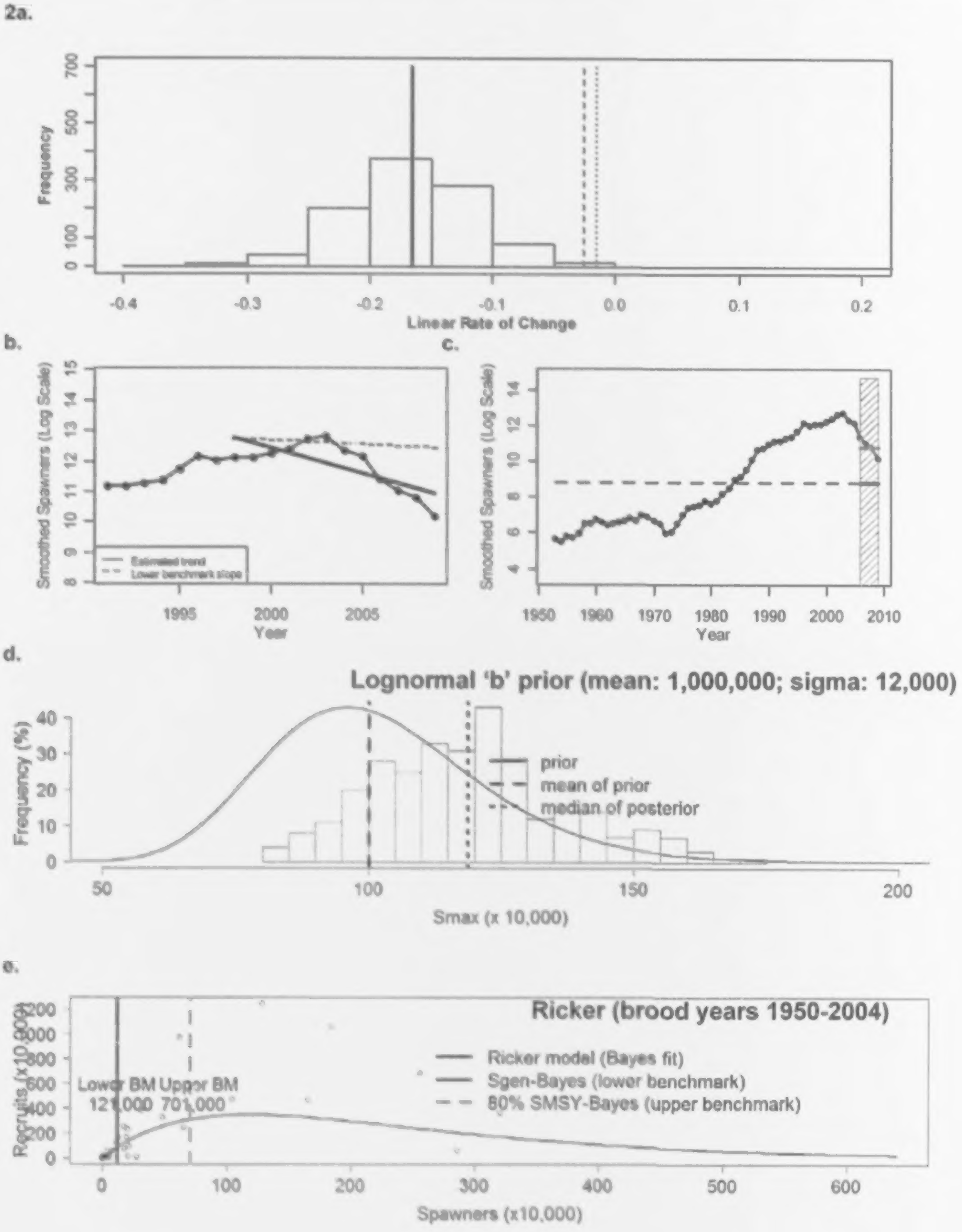
e.



Ricker (brood years 1950-2004)
Ricker model (Bayes fit)
Sgen-Bayes (lower benchmark)
80% SMSY-Bayes (upper benchmark)
Lower BM
Upper BM
6,000
22,000
Recruits (x10,000)
5
10
15
20
0
5
10
15
20
25
Spawners (x10,000)

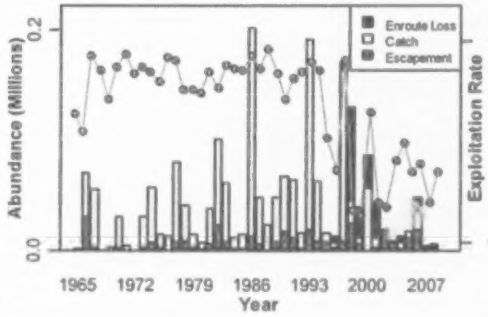
Quesnel-S


1a.
Enroute Loss
Catch
Escapement
Abundance (Millions)
0
2
4
6
8
10
12
Exploitation Rate
0
1
1950
1959
1968
1977
1986
1995
2004
Year
b.
Female Pre-spawn Mortality
Effective Female Spawners
Male Spawners
0.0
1.0
2.0
3.0
1950
1959
1968
1977
1986
1995
2004
Year
c.
Standardized Productivity Indices
2
0
-2
-4
ln(R/EFS)
Ricker Residuals
Kalman Filter
1950
1960
1970
1980
1990
2000
2010
Brood Year
d.
Recruits per Spawner
(Log Scale)
4
2
0
-2
1950
1960
1970
1980
1990
2000
2010
Brood Year
e.
Freshwater Survival
(fall-fry/EFS)
0
200
400
600
800
1000
1950
1960
1970
1980
1990
2000
2010
Brood Year
f.
Post-Fry Survival
(recruits/fall-fry)
0.0
0.2
0.4
0.6
1950
1960
1970
1980
1990
2000
2010

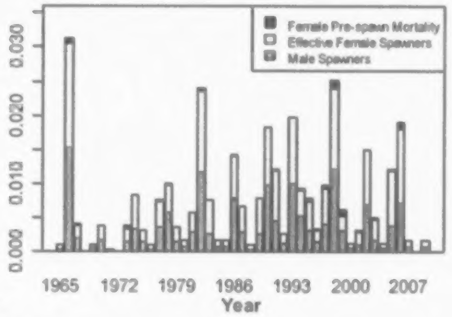
2a.
Frequency
0
100
300
500
700
-0.4
-0.3
-0.2
-0.1
0.0
0.1
0.2
Linear Rate of Change
b.
Smoothed Spawners (Log Scale)
8
9
10
11
12
13
14
15
Estimated trend
Lower benchmark slope
1995
2000
2005
Year
c.
Smoothed Spawners (Log Scale)
4
6
8
10
12
14
1950
1960
1970
1980
1990
2000
2010
Year
d.
Lognormal 'b' prior (mean: 1,000,000; sigma: 12,000)
Frequency (%)
0
10
20
30
40
prior
mean of prior
median of posterior
50
100
150
200
Smax (x 10,000)
e.
Ricker (brood years 1950-2004)
Recruits (x10,000)
0
200
400
600
800
1000
1200
Ricker model (Bayes fit)
Sgen-Bayes (lower benchmark)
80% SMSY-Bayes (upper benchmark)
Lower BM
121,000
Upper BM
701,000
0
100
200
300
400
500
600
Spawners (x10,000)

Seton-L (*de novo*)

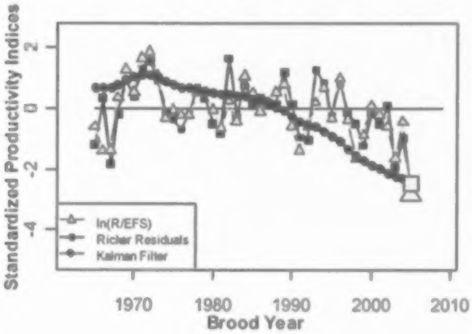
1a.



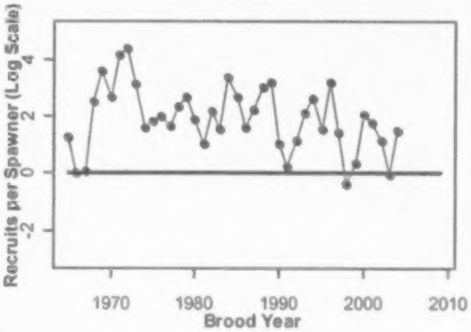
b.



c.



d.

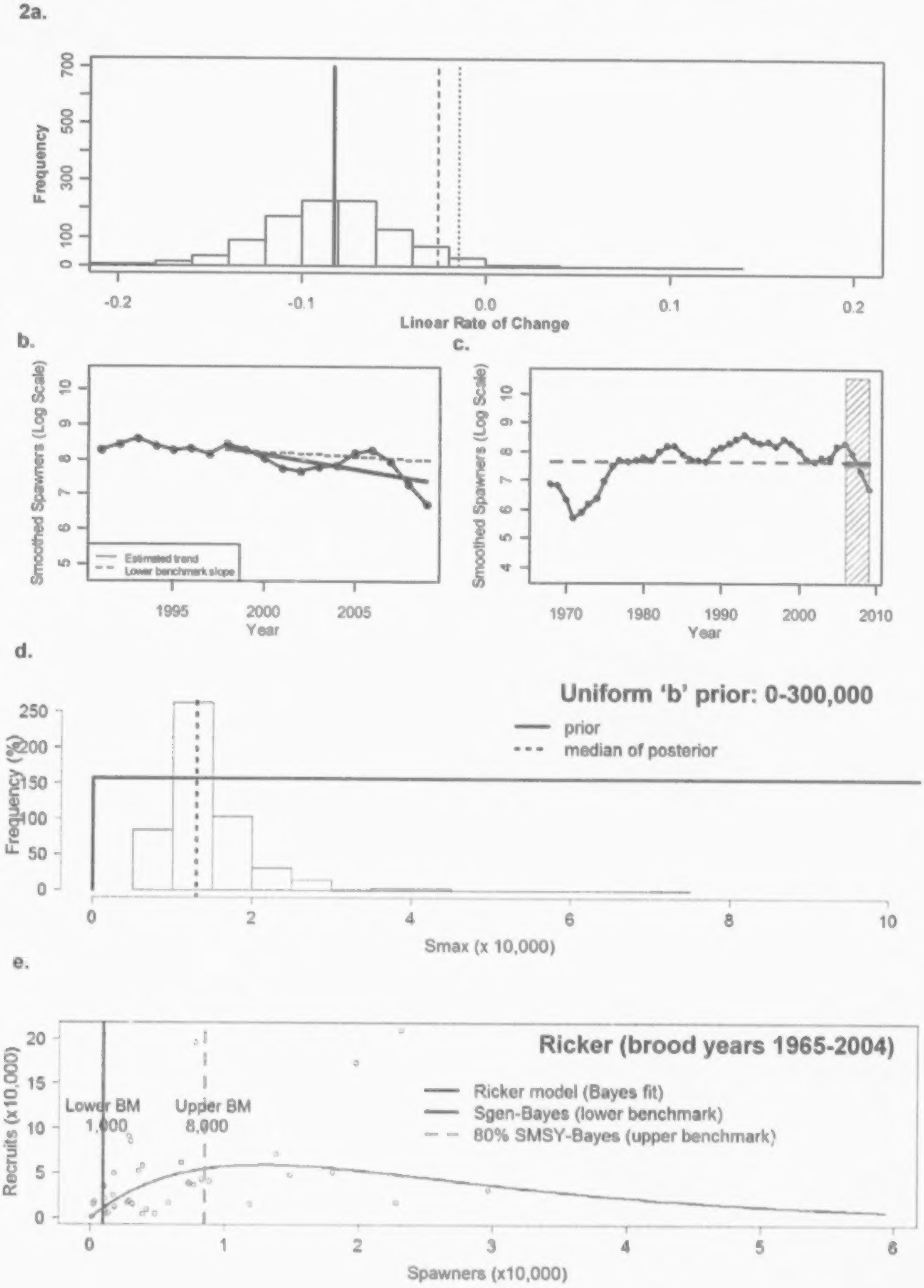


e.

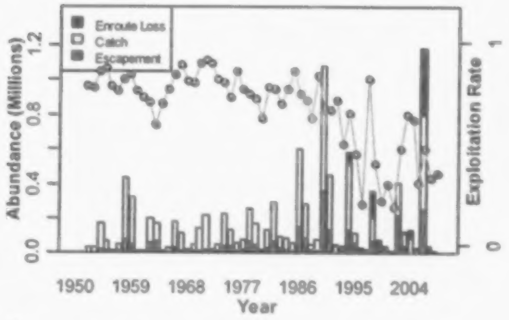
No freshwater survival data are available

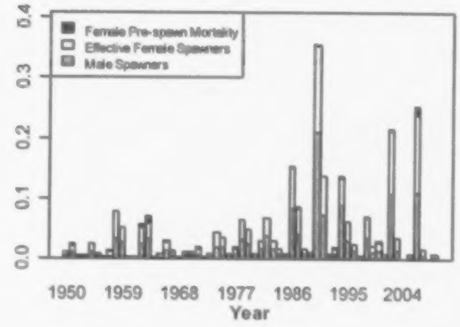
f.

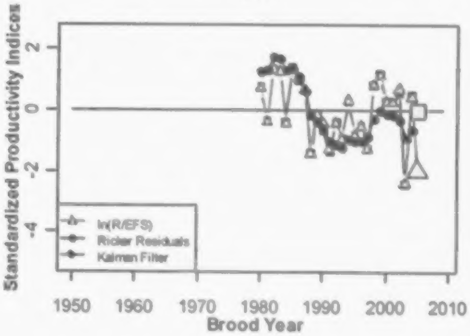
No marine survival data are available

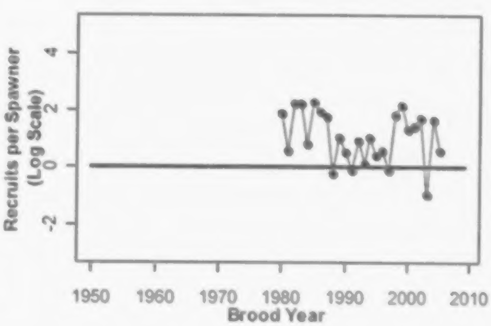

2a.
Frequency
0
100
300
500
700
-0.2
-0.1
0.0
0.1
0.2
Linear Rate of Change
b.
Smoothed Spawners (Log Scale)
5
6
7
8
9
10
Estimated trend
Lower benchmark slope
1995
2000
2005
Year
c.
Smoothed Spawners (Log Scale)
4
5
6
7
8
9
10
1970
1980
1990
2000
2010
Year
d.
Uniform 'b' prior: 0-300,000
prior
median of posterior
Frequency (%)
0
50
100
150
200
250
0
2
4
6
8
10
Smax (x 10,000)
e.
Ricker (brood years 1965-2004)
Ricker model (Bayes fit)
Sgen-Bayes (lower benchmark)
80% SMSY-Bayes (upper benchmark)
Lower BM
1,000
Upper BM
8,000
Recruits (x10,000)
0
5
10
15
20
0
1
2
3
4
5
6
Spawners (x10,000)

Shuswap-ES





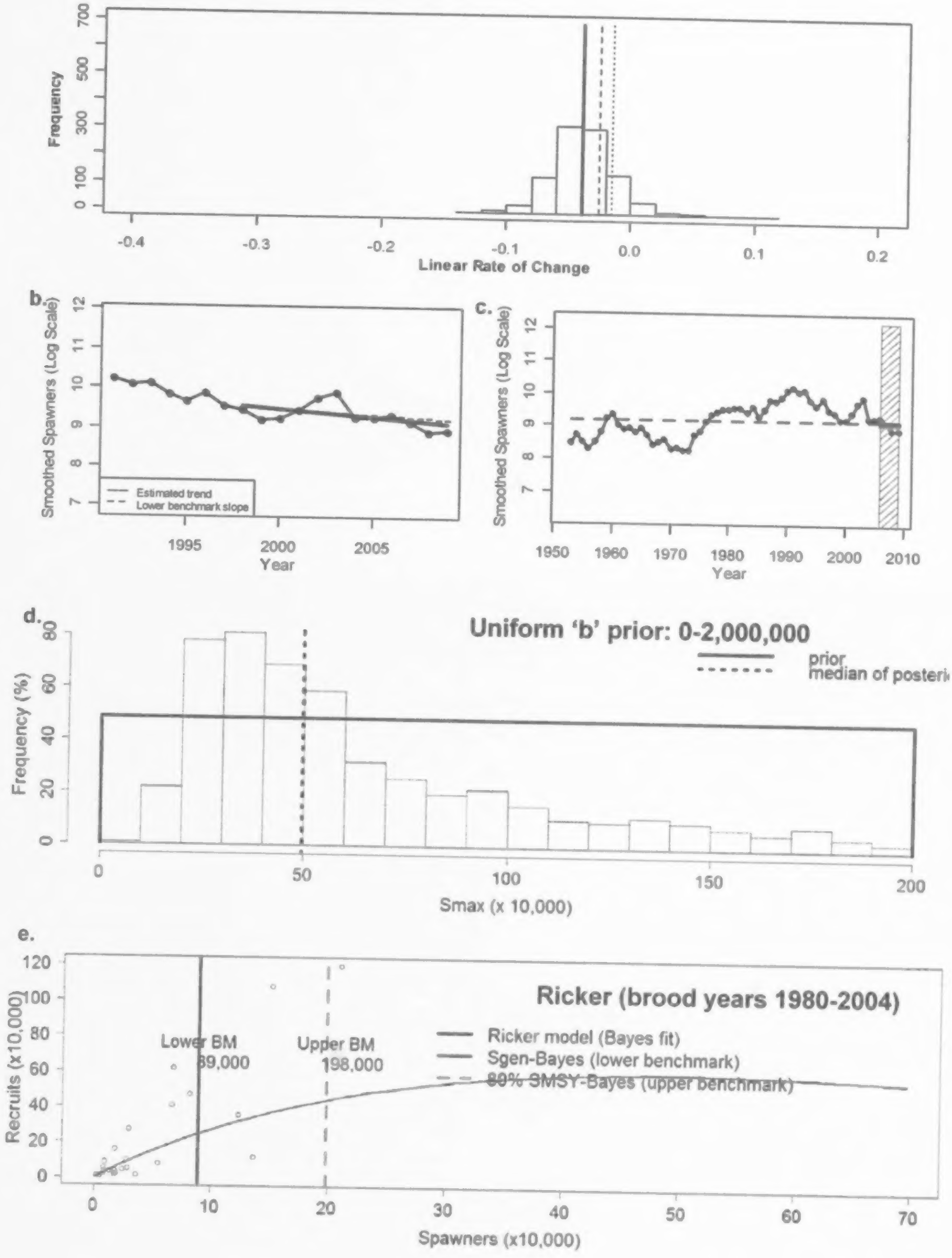




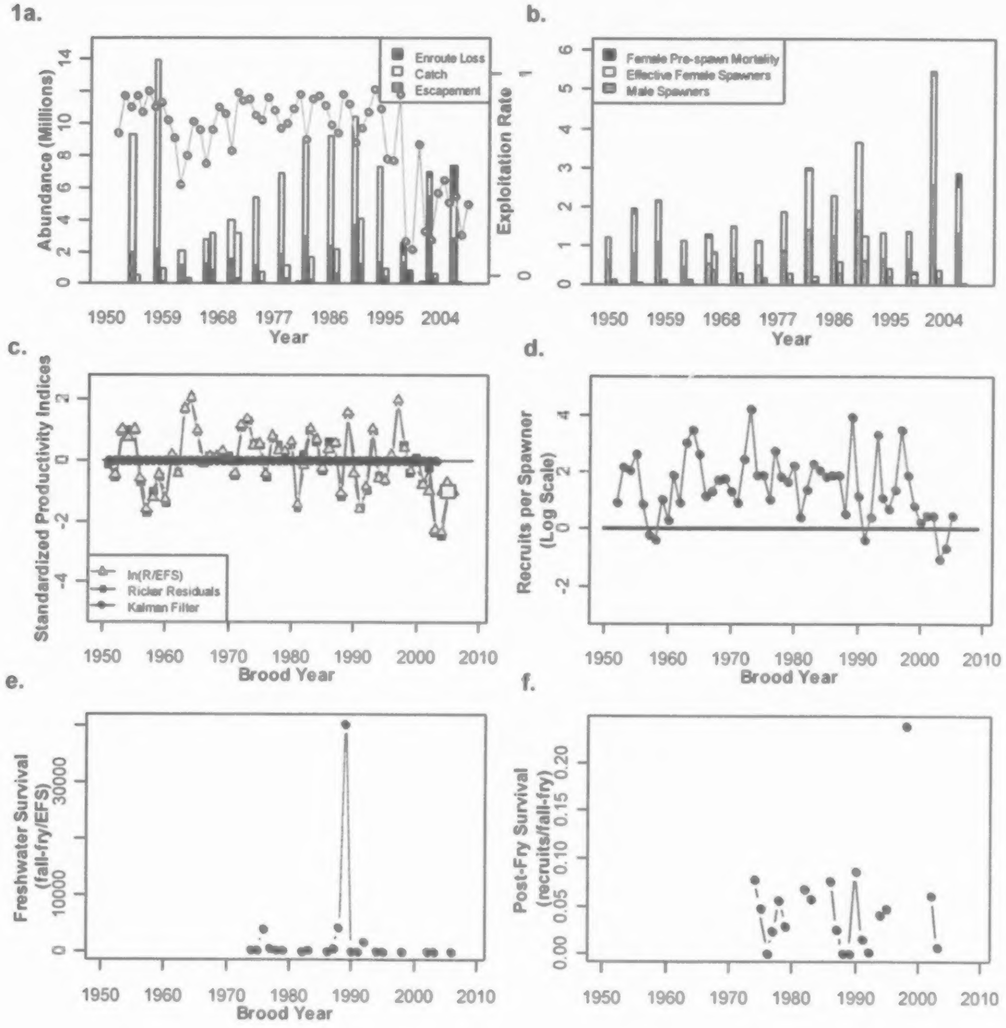
No freshwater survival data are available

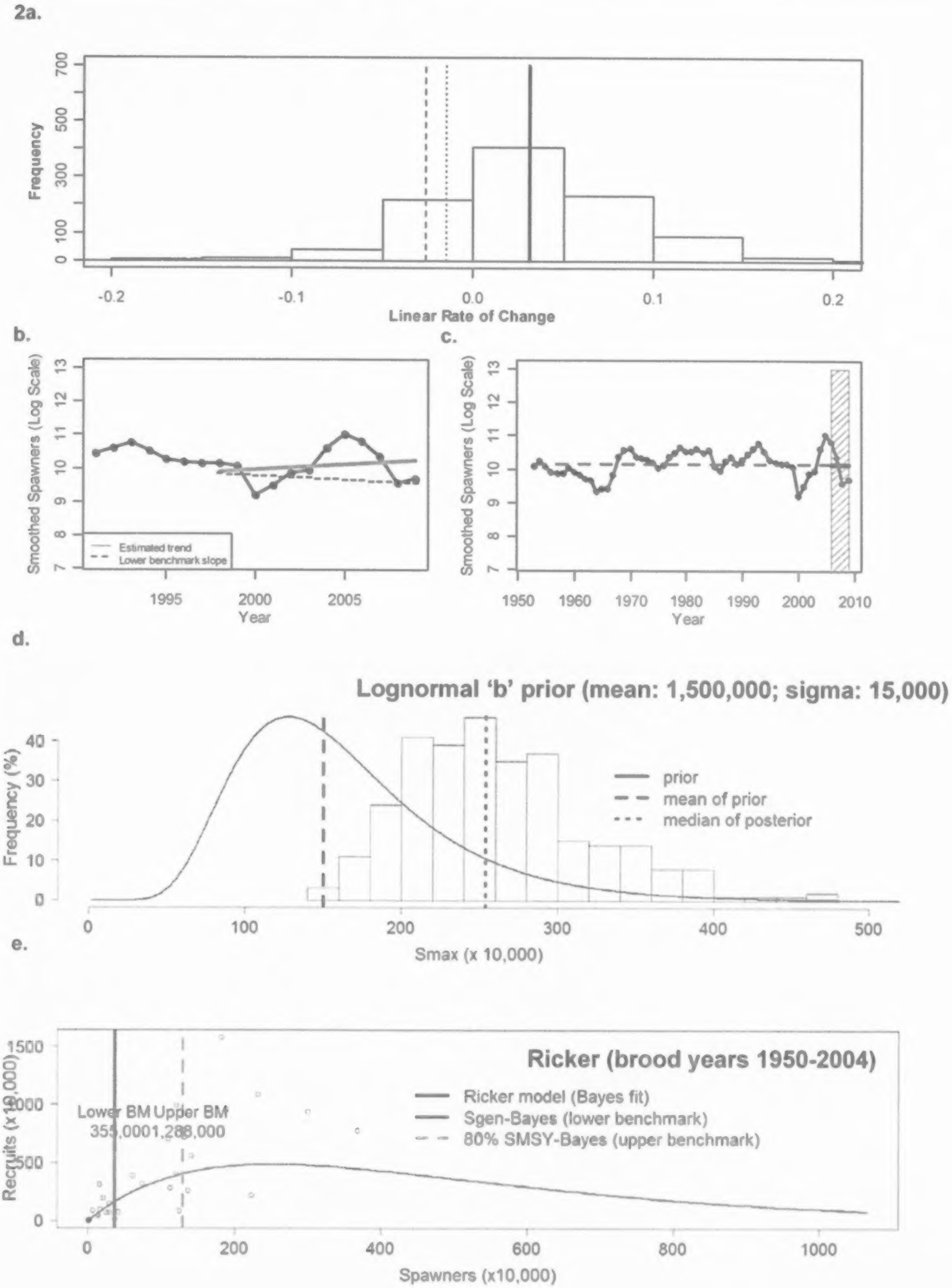
No marine survival data are available

2a.

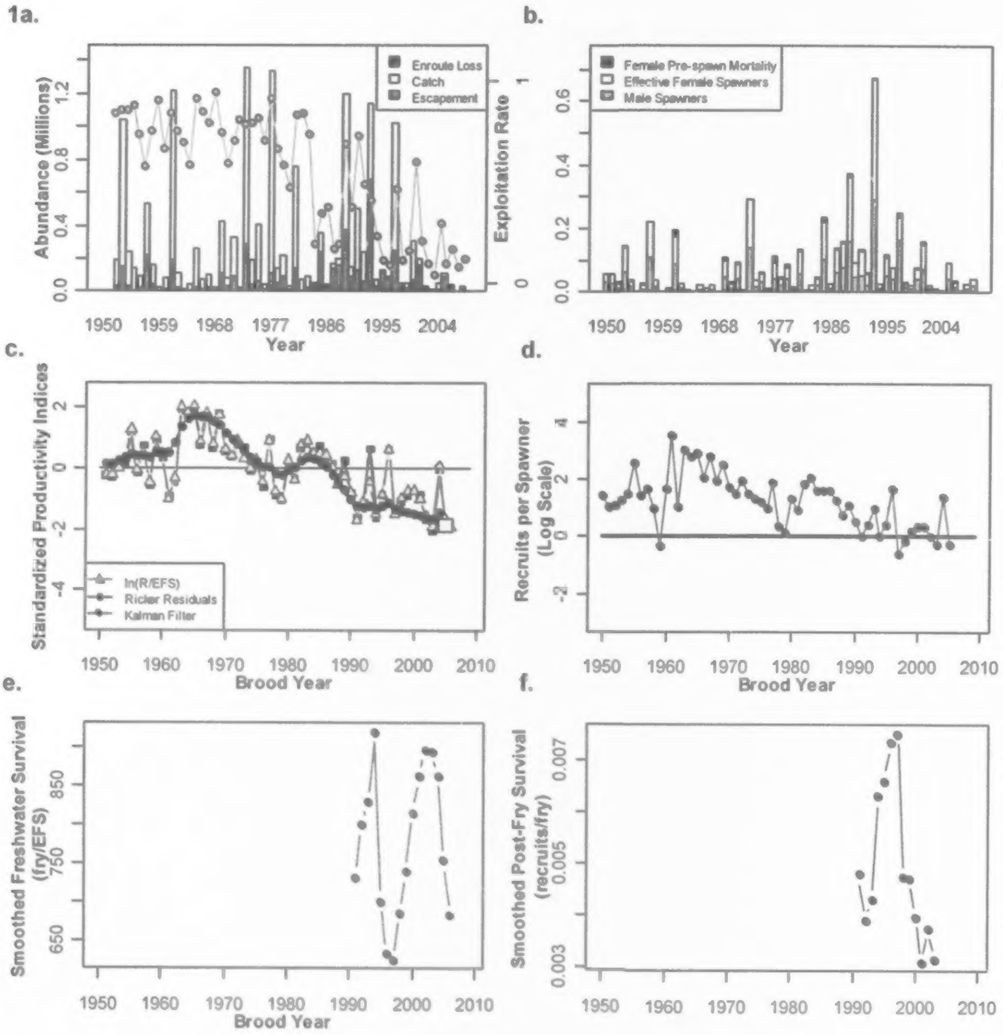

Frequency
0
100
300
500
700
-0.4
-0.3
-0.2
-0.1
0.0
0.1
0.2
Linear Rate of Change
b.
Smoothed Spawners (Log Scale)
7
8
9
10
11
12
Estimated trend
Lower benchmark slope
1995
2000
2005
Year
c.
Smoothed Spawners (Log Scale)
7
8
9
10
11
12
1950
1960
1970
1980
1990
2000
2010
Year
d.
Uniform 'b' prior: 0-2,000,000
prior
median of posteri
Frequency (%)
0
20
40
60
80
0
50
100
150
200
Smax (x 10,000)
e.
Ricker (brood years 1980-2004)
Ricker model (Bayes fit)
Sgen-Bayes (lower benchmark)
80% SMSY-Bayes (upper benchmark)
Lower BM
89,000
Upper BM
198,000
Recruits (x10,000)
0
20
40
60
80
100
120
0
10
20
30
40
50
60
70
Spawners (x10,000)

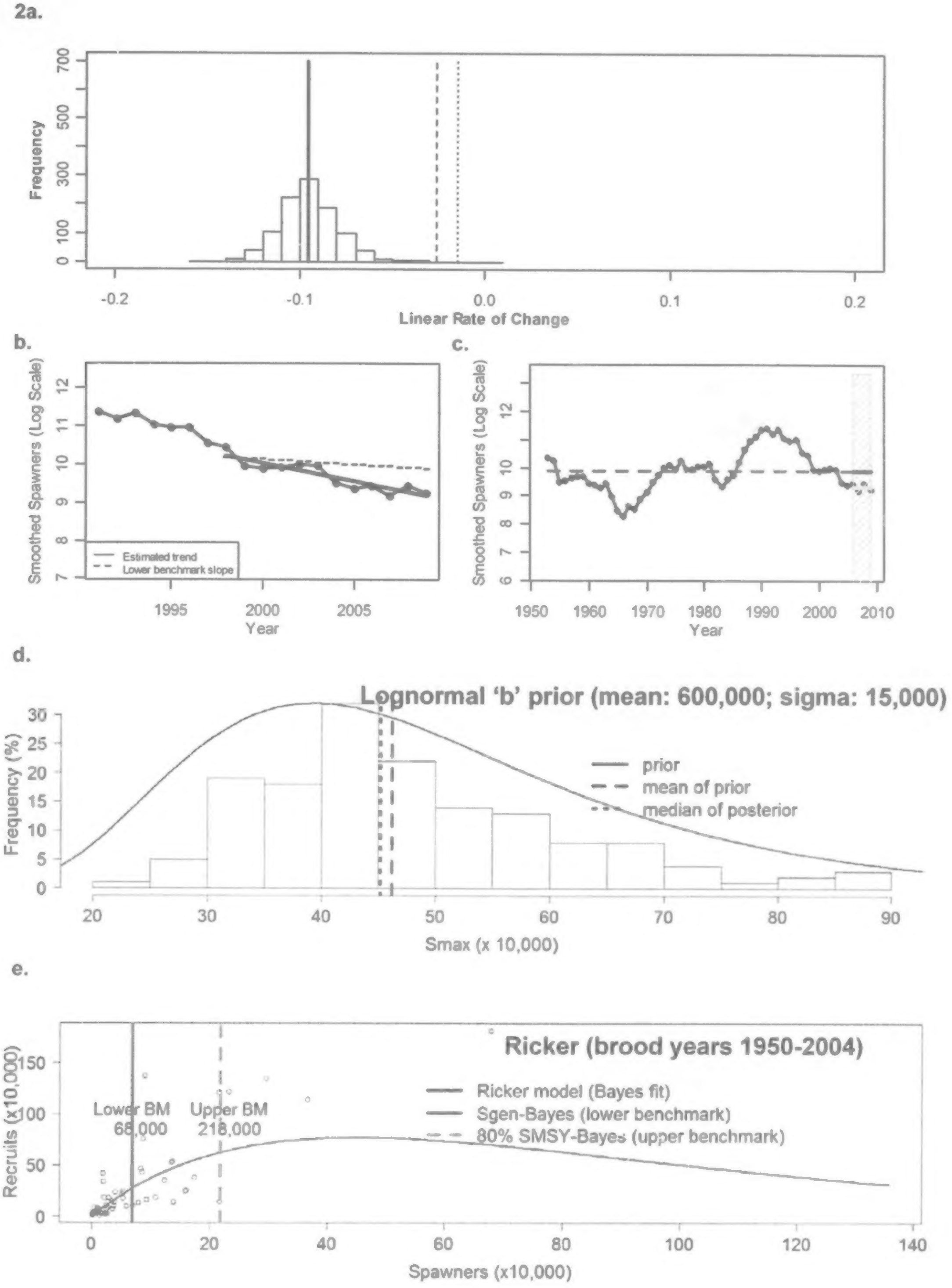
Shuswap Complex-L




2a.
Frequency
0
100
300
500
700
-0.2
-0.1
0.0
0.1
0.2
Linear Rate of Change
b.
Smoothed Spawners (Log Scale)
7
8
9
10
11
12
13
Estimated trend
Lower benchmark slope
1995
2000
2005
Year
c.
Smoothed Spawners (Log Scale)
7
8
9
10
11
12
13
1950
1960
1970
1980
1990
2000
2010
Year
d.
Lognormal 'b' prior (mean: 1,500,000; sigma: 15,000)
Frequency (%)
0
10
20
30
40
prior
mean of prior
median of posterior
0
100
200
300
400
500
Smax (x 10,000)
e.
Ricker (brood years 1950-2004)
Recruits (x 10,000)
0
500
1000
1500
Lower BM
355,000
Upper BM
1,288,000
Ricker model (Bayes fit)
Sgen-Bayes (lower benchmark)
80% SMSY-Bayes (upper benchmark)
0
200
400
600
800
1000
Spawners (x10,000)

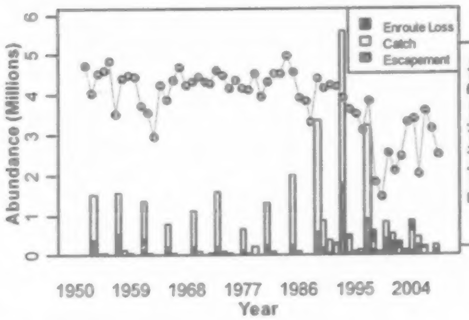
Takla-Trembleur-Early Stuart (EStu)



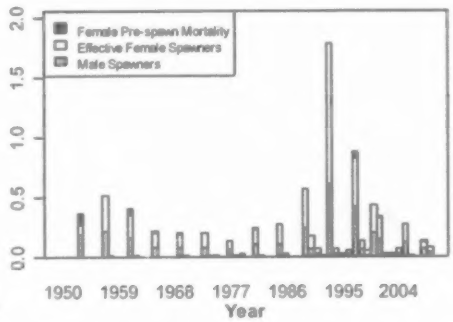
2a.
Frequency
0
100
300
500
700
-0.2
-0.1
0.0
0.1
0.2
Linear Rate of Change
b.
Smoothed Spawners (Log Scale)
7
8
9
10
11
12
Estimated trend
Lower benchmark slope
1995
2000
2005
Year
c.
Smoothed Spawners (Log Scale)
6
7
8
9
10
12
1950
1960
1970
1980
1990
2000
2010
Year
d.
Lognormal 'b' prior (mean: 600,000; sigma: 15,000)
Frequency (%)
0
5
10
15
20
25
30
prior
mean of prior
median of posterior
20
30
40
50
60
70
80
90
Smax (x 10,000)
e.
Ricker (brood years 1950-2004)
Recruits (x10,000)
0
50
100
150
Lower BM
68,000
Upper BM
218,000
Ricker model (Bayes fit)
Sgen-Bayes (lower benchmark)
80% SMSY-Bayes (upper benchmark)
0
20
40
60
80
100
120
140
Spawners (x10,000)

Takla-Trembleur-Stuart-S

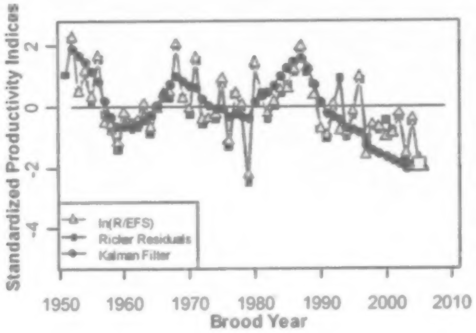
1a.



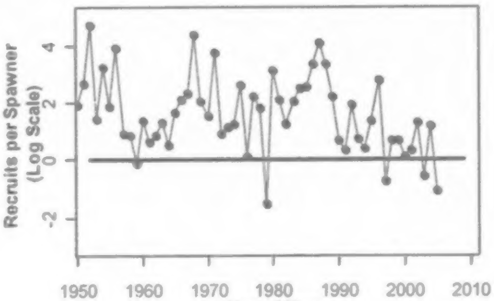
b.



c.



d.

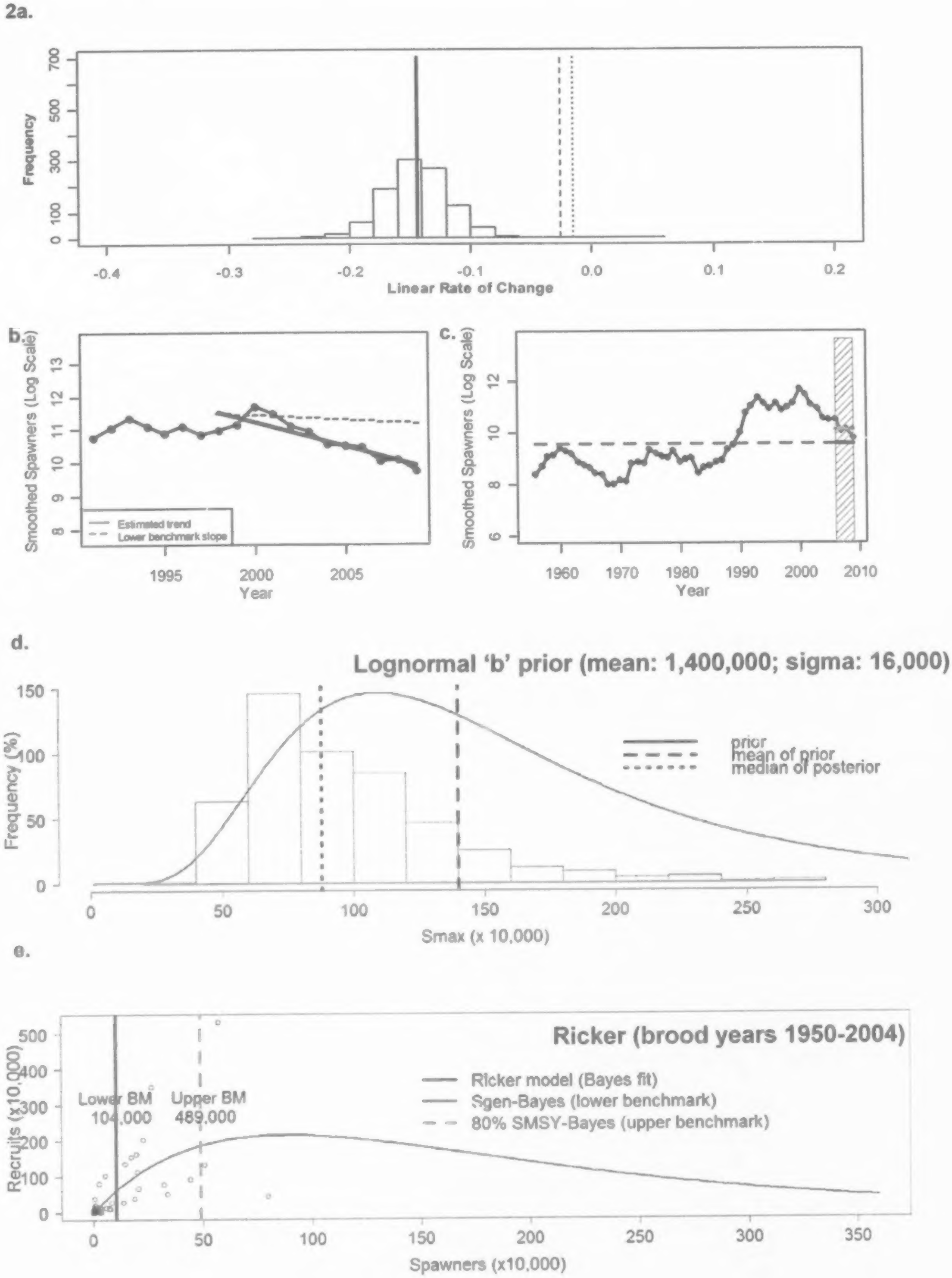


e.

No freshwater survival data are available

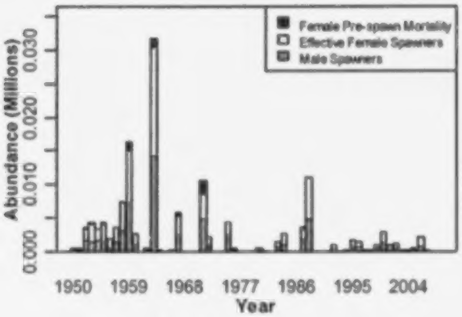
f.

No marine survival data are available


2a.
Frequency
0
100
300
500
700
-0.4
-0.3
-0.2
-0.1
0.0
0.1
0.2
Linear Rate of Change
b.
Smoothed Spawners (Log Scale)
8
9
10
11
12
13
Estimated trend
Lower benchmark slope
1995
2000
2005
Year
c.
Smoothed Spawners (Log Scale)
6
8
10
12
1960
1970
1980
1990
2000
2010
Year
d.
Lognormal 'b' prior (mean: 1,400,000; sigma: 16,000)
Frequency (%)
0
50
100
150
prior
mean of prior
median of posterior
0
50
100
150
200
250
300
Smax (x 10,000)
e.
Ricker (brood years 1950-2004)
Recruits (x10,000)
0
100
200
300
400
500
Lower BM
104,000
Upper BM
489,000
Ricker model (Bayes fit)
Sgen-Bayes (lower benchmark)
80% SMSY-Bayes (upper benchmark)
0
50
100
150
200
250
300
350
Spawners (x10,000)

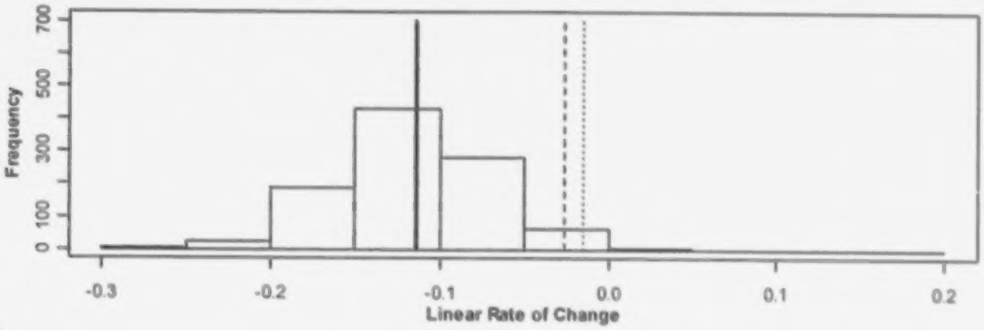
Taseko-ES

1b.

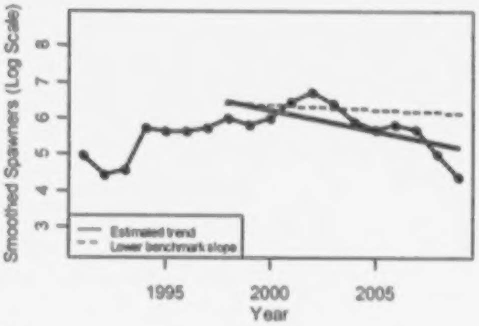


Only escapement data are available for Taseko-ES.

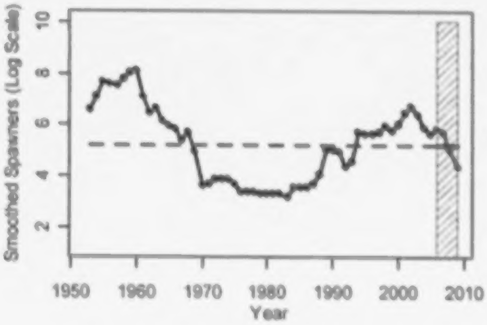
2a.



b.



c.



d.

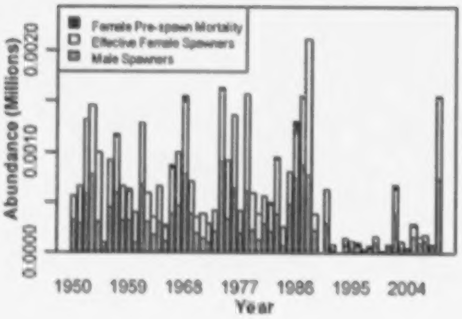
No stock-recruitment data are available to estimate abundance based benchmarks

e.

No stock-recruitment data are available to estimate abundance based benchmarks

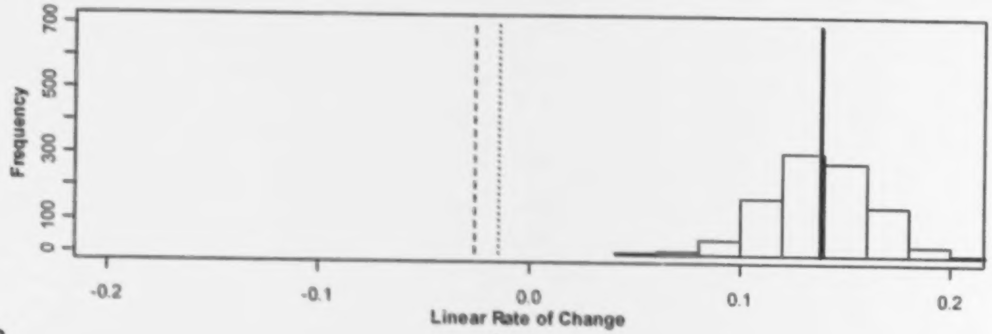
Widgeon-(River-Type)

1b.

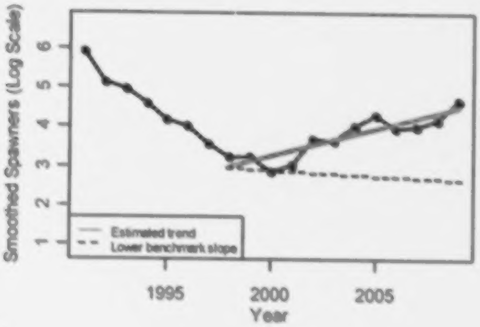


Only escapement data are available for Widgeon (River-Type).

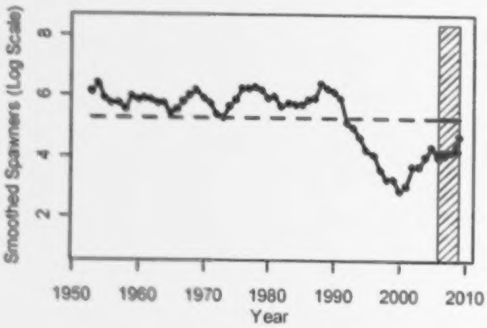
2a.



b.



c.



d.

No stock-recruitment data are available to estimate abundance based benchmarks

e.

No stock-recruitment data are available to estimate abundance based benchmarks

APPENDIX 4: Updated spawner abundance that maximizes juvenile production (S_{max}) for lakes assessed in the Fraser Watershed.

Table A. Summary of trawl catch for each survey used to estimate juvenile Sockeye competitor biomass for the PR model by lake. Empty cells indicate no fish from the competitor category (identified in column headings) were caught in the trawl survey.

Lake	Year	Survey	DNA/ otolith	Age-0 nerka	Age-1 nerka	Age-2+ nerka	Age-0 other	Other large fish
Adams	1997	199714	y	160				
Adams	1998	199811	y	275				
Anderson	2000	200010	y	496	27	1	1	1
Anderson	2001	200107	n	337	24			
Anderson	2002	200209	n	95	9	1		1
Anderson	2003	200308	n	150	34	8		
Bowron	2004	200406	n	134		1	2	
Chilliwack	2001	200110	n	509	5	3		
Chilliwack	2002	200212	y	10	3	1		
Chilliwack	2009	200905	y	94	2			
Cultus	2001	200109	n	2		1	7	
Cultus	2002	200211	n				6	
Cultus	2009	200901	n		56	1	53	1
Fraser	1992	199205	n	152	1		4	
Harrison	1999	199910	n	324			2,737	1
Lillooet	2000	200011	n	60	1		7	1
Quesnel	1987	198703	n	323	13	3		
Quesnel	1988	198808	n	17	2	3		
Quesnel	1994	199404	n	684		2		
Quesnel	2003	200306	n	1,252	7	1		
Quesnel	2004	200407	n	637	1			
Seton	2000	200008	y	40	3	60		
Seton	2001	200108	n	146	1	2	1	
Seton	2002	200208	n	230	18			
Shuswap	1987	198702	n	2,780	1		1	1
Shuswap	1988	198814	n	1,124	56	14	1	
Shuswap	1989	198914	n	160		2		
Shuswap	1990	199019	n	1,111	16	6	5	
Shuswap	1991	199117	n			1		
Stuart	1996	199607	y	489		22		
Stuart	1997	199709	y	443			8	
Stuart	1998	199808	y	189		11	6	
Takla	1996	199605	y	292	16	19		1
Takla	1997	199710	y	230	21	2		
Takla	1998	199809	y	657	5	1	4	
Trembleur	1996	199606	y	226		2	2	
Trembleur	1997	199712	y	238			2	
Trembleur	1998	199805	y	861				

Table B. Updated biomass estimates (kg/lake) by competitor category for each survey used to estimate competitor biomass for the PR model. Note that n/a indicates some unknown quantity of competitor biomass that could not be estimated from the trawl and acoustic data.

Lake	Year	Survey	Age-0 kokanee	Age-1 kokanee	Age-2+ kokanee	Other age-0	Total biomass
Adams	1997	199714	3,235	0	0	0	3,235
Adams	1998	199811	694	0	0	0	694
Anderson	2000	200010	1,421	416	9,543	0	11,381
Anderson	2001	200107	n/a	431	0	0	431
Anderson	2002	200209	n/a	653	6,248	0	6,901
Anderson	2003	200308	n/a	1,120	45,536	0	46,656
Bowron	2004	200406	n/a	n/a	n/a	9	9
Chilliwack	2001	200110	n/a	n/a	10,402	n/a	10,402
Chilliwack	2002	200212	730	n/a	1,048	n/a	1,777
Chilliwack	2009	200905	1,502	n/a	n/a	0	1,502
Cultus	2001	200109	n/a	0	n/a	113	113
Cultus	2002	200211	n/a	0	0	562	562
Cultus	2009	200901	n/a	n/a	0	190	190
Fraser	1992	199205	n/a	3,829	0	n/a	3,829
Harrison	1999	199910	n/a	0	0	30,376	30,376
Lillooet	2000	200011	n/a	34	0	0	34
Quesnel	1987	198703	n/a	39,592	n/a	0	39,592
Quesnel	1988	198808	n/a	26,144	n/a	0	26,144
Quesnel	1994	199404	n/a	0	n/a	n/a	0
Quesnel	2003	200306	n/a	n/a	45,931	0	45,931
Quesnel	2004	200407	n/a	188	0	0	188
Seton	2000	200008	675	1,288	12,113	0	14,075
Seton	2001	200108	n/a	n/a	27,611	0	27,611
Seton	2002	200208	n/a	3,854	0	0	3,854
Shuswap	1987	198702	n/a	4,448	0	0	4,448
Shuswap	1988	198814	n/a	82,527	n/a	0	82,527
Shuswap	1989	198914	n/a	9,558	n/a	n/a	9,558
Shuswap	1990	199019	n/a	n/a	n/a	0	0
Shuswap	1991	199117	n/a	0	n/a	0	0
Stuart	1996	199607	24,307	0	90,419	0	114,726
Stuart	1997	199709	42,288	0	0	907	43,195
Stuart	1998	199808	19,086	0	n/a	808	19,894
Takla	1996	199605	6,723	5,582	52,177	0	64,483
Takla	1997	199710	5,771	13,436	81,814	0	101,022
Takla	1998	199809	6,178	1,441	134,714	23	142,357
Trembleur	1996	199606	11,926	0	n/a	31	11,957
Trembleur	1997	199712	3,830	0	0	8	3,838
Trembleur	1998	199805	202	0	0	0	202

Table C. Preliminary mean PR model estimates of the Fraser River Sockeye productive capacity of various rearing lakes. PR data are based on monthly sampling of May-October growing season for 1 or more years, with the exception of Pitt Lake. The presence of age-2 smolts has been accounted for in Chilko and Cultus lakes only. (*) indicates lakes for which competitor biomass was extrapolated from similar lakes. Escapement is in total adult spawners.

				Unadjusted PR model predictions				Adjusted PR model predictions		
Lake	Comment	Mean seasonal PR (mg C /m^2)	PR$_{total}$ (t C/lake)	Smolt biomass (kg) (R_{max})	Smolt #'s	Escapement (S_{max})	Prop. of PR$_{total}$ used by competitor biomass	Smolt biomass (kg) (R_{max})	Smolt #'s	Escapement (S_{max})
Adams	Probably affected by fertilization	115	2659	120,970	26,882,310	497,175	6%	113,712	25,269,371	466,934
Anderson	Mean all years	303	1527	69,484	15,440,880	285,571	37%	43,775	9,727,754	179,752
Bowron	2004 only	131	219	9,947	2,210,536	40,883	0%	9,947	2,210,536	40,847
Chilko*	Fertilized Mean	103	3396	154,539	34,341,944	635,137	0%	154,539	29,556,995	546,162
Chilko*	Natural mean ?1995	69	2295	104,432	23,207,184	429,205	0%	104,432	21,335,362	394,240
Chilko*	2009 natural	121	4020	182,922	40,649,286	751,788	0%	182,922	37,370,636	690,544
Chilliwack	3 year mean	101	218	9,926	2,205,840	40,796	37%	6,254	1,389,679	25,679
Cultus	3 year mean	404	457	20,779	4,617,558	85,399	6%	19,532	4,316,524	79,762
Francois*	2 year mean	163	7247	329,738	73,275,020	1,355,185	0%	329,738	73,275,020	1,353,995
Fraser	2 year mean	332	3227	146,830	32,628,960	603,456	6%	138,021	30,671,222	566,751
Harrison	2 year mean	109	4336	197,289	43,841,980	810,836	37%	124,292	27,620,447	510,378
Kamloops*	2007	257	2378	108,188	24,041,836	444,642	0%	108,188	24,041,836	444,251
Lillooet	2000	163	880	40,022	8,893,783	164,486	0%	40,022	8,893,783	164,342
Mabel*	2 year mean	203	2160	98,285	21,841,092	403,940	6%	92,388	20,530,626	379,370
Pitt*	Jul, Oct 1989 & Mar 1990	72	617	28,056	6,234,608	115,306	37%	17,675	3,927,803	72,579
Quesnel	Mean all 10 years	125	6075	276,413	61,425,000	1,136,025	6%	259,828	57,739,500	1,066,926
Quesnel	Pre- 1995 mean (5 yrs)	104	5054	229,975	51,105,600	945,173	6%	216,177	48,039,264	887,682
Quesnel	Post 2003 mean (5 yrs)	130	6318	287,469	63,882,000	1,181,466	6%	270,221	60,049,080	1,109,603
Seton	4 year mean	233	1007	45,798	10,177,440	188,227	37%	28,853	6,411,787	118,479
Shuswap	6 year mean	171	10159	462,252	102,722,620	1,899,804	6%	434,517	96,559,263	1,784,247
Stuart	3 year mean	137	8899	404,914	89,980,800	1,664,150	37%	255,096	56,687,904	1,047,494
Takla	3 year mean	56	2475	112,624	25,027,475	462,871	37%	70,953	15,767,309	291,352
Trembleur	3 year mean	84	1769	80,491	17,886,960	330,810	6%	75,662	16,813,742	310,689

APPENDIX 5: Methodology used for gap filling CU time series data where required.

Cycle-Line Average Method

Application: CUs with only one site or for CUs with multiple sites where no sites have abundance estimates in a given year.

Method: Missing values are interpolated using the average of the escapement estimates for the previous and subsequent generation on that cycle. In cases where the previous and subsequent estimates are not available, the average of up to two generations away from the gap is used; if no data are available within two generations of the gap, the gap is assumed to equal zero (usually systems are not assessed when abundance is assumed negligible) or the years are not included in the time series (in most cases large gaps occur in the early time series). Interpolation was conducted prior to $\log_e$ transformation and smoothing with a four year running average.

Example: **Lillooet-Harrison-L**
Birkenhead was gap filled for the 2002 estimate. Birkenhead is the only site used for this CU, therefore gaps were filled using this cycle-line average method. The 2002 gap was filled using the average of the previous generation (1998 EFS estimate: 172,997) and the subsequent generation (2006 EFS estimate: 137,365), producing a gap-filled estimate of 155,181 EFS for the 2002 year.

Usage: Lillooet-Harrison-L, Taseko-ES, Widgeon (River-Type)-L

Mean Proportion Method

Application: CUs with multiple streams

Method: This method of gap filling is based on the assumption of spatial correlation between sites. The method uses trends in the escapement time series' of spatially related stream aggregates to interpolate missing values for individual streams within that aggregate. Each CU was used as an aggregate, assuming that trends in escapement were consistent across streams in a CU. One exception to this was the very large Takla-Trembleur-EStu CU, in which individual sites and groups of sites exhibited very different trends. For the Takla-Trembleur-EStu CU, therefore, sites were grouped into six separate aggregates based on their location in the watershed and correlation in abundance trends.

This method calculates the mean abundance of each stream across the years of available data, including only years for which all streams in the aggregate had recorded data. This was to account for possible changes in the escapement trend in years in which streams had missing data, ensuring that the proportion calculations were representative.

$\bar{E}_s = \frac{\sum_{y=1}^{Y} E_{sy}}{Y_s}$, where $\bar{E}_s$ is the mean escapement for a CU(s), E_{sy} is recorded escapement for each CU (*s*), y = years with escapement data for all streams, Y_s = total number of years with escapement data for all CUs. The proportion that each site contributes to the CU over the course of the time series is calculated as: $P_s = \frac{\bar{E}_s}{\sum_{s=1}^{S} \bar{E}_s}$, where P_s is the proportion the site (*s*) comprises of the total CU aggregate. S = the total number of stocks in aggregate *a*. Expansion factors are then calculated for each year of aggregate data in order to expand the aggregate to account for missing stocks in each year, $F_y = \frac{1}{\sum_{s=1}^{S} P_{sy}}$, where F_y is the expansion factor for each year in an aggregate and P_{sy} is the proportion contributed for each stream in that year (missing values will = 0). Finally, the new aggregate sum for each year is calculated as the product of the expansion factor and the sum of the recorded escapement data across streams: $E'_y = F_y * \sum_{s=1}^{S} E_{sy}$, where E'_y is the expanded aggregate, and E_{sy} is the recorded escapement of each stream in that year.

Example: Nahatlatch-ES had missing data for the Nahatlatch Lake site in 1975, 1976, and 1978. The average escapements for both Nahatlatch Lake and River were calculated excluding these years from the dataset, resulting in proportional contributions of 0.25 and 0.75, respectively, to the Nahatlatch CU. [When the entire dataset is used, the proportions are 0.26 and 0.74, because the low escapements to Nahatlatch River in 1975, 1976 and 1978 are included in the average, while the Nahatlatch Lake average is not being pulled down by these low years.]

Usage: Nahatlatch-ES, Shuswap-ES, Takla-Trembleur-Stuart-S

Mean Proportion Method- Cyclic (Dominant/Sub-dominant or all cycles)

Application: In highly cyclic CUs, where the dominant and (in some cases) sub-dominant cycles are highly different from both each other and the off-cycle years in term of abundance.

Method: For highly cyclic CUs, the proportional contribution of individual sites tends to differ between cycle years. Therefore, the average escapement and the site proportions were calculated individually for each cycle, in order to be representative of actual patterns when gap-filling.

Example: In the Shuswap Complex-L CU, the Adams River site contributes 71% of the spawning escapement, on average, in dominant cycle years, whereas in subdominant years this site represents 95% of Shuswap Complex-L escapement.

Usage: Shuswap Complex-L, Takla-Trembleur-EStu, Quesnel-S

APPENDIX 6: Pacific Science Advisory Review Committee (PSARC) Request for Wild Salmon Policy Stock Status Evaluation for Fraser Sockeye

REQUEST FOR SCIENCE INFORMATION AND/OR ADVICE

PART 1: **DESCRIPTION OF THE REQUEST – TO BE FILLED BY THE CLIENT REQUESTING THE INFORMATION/ADVICE**

Date *(when initial client's submission is sent to Science) (dd/mm/yyyy)*:

Directorate, Branch or group initiating the request and category of request

Directorate/Branch/Group
- ☒ Fisheries and Aquaculture Management
- ☐ Oceans & Habitat Management and SARA
- ☐ Policy
- ☒ Science
- ☐ Other (please specify):

Category of Request
- ☒ Stock Assessment
- ☐ Species at Risk
- ☐ Human impacts on Fish Habitat/ Ecosystem components
- ☐ Aquaculture
- ☐ Ocean issues
- ☐ Invasive Species
- ☐ Other (please specify):

Initiating Branch Contact:

Name: Paul Ryall (Lead, Salmon Team)
Email: Paul.Ryall@dfo-mpo.gc.ca
Telephone Number: 604-666-0115
Fax Number: 604-666-9136

Issue Requiring Science Advice (i.e., "the question"):
Issue posed as a question for Science response.

1. Develop Wild Salmon Policy (WSP) lower benchmarks for up to 36 Fraser Sockeye WSP Conservation Units (CUs) where data availability permits; several of these 36 CUs have been flagged by Fisheries and Oceans Canada (DFO) Stock Assessment as being opportunistic spawning sites only rather than CUs. For each CU, up to four broad criteria (abundance, temporal trends in abundance, distribution of spawners, and fishing mortality) may be used for benchmark development depending on data quality and availability. The total number of lower benchmarks for each CU will vary depending on the criteria and associated benchmarks used; each criteria used could have more than one benchmark. The first step before identifying lower benchmarks on spawner abundances specifically will require the compilation/estimation of the recruitment time series by CU and subsequently the estimation of stock-recruitment parameters.

2. Provide a preliminary assessment of stock status for all Fraser Sockeye CUs using the WSP lower benchmarks. This step will be an iterative process as it is amongst the first salmon group in the Pacific Region where WSP lower benchmarks are being developed; not all methodology has been finalized including the use of multiple benchmarks to assess status.

Rationale for Advice Request:
What is the issue, what will it address, importance, scope and breadth of interest, etc.?

The development of Wild Salmon Policy (WSP) benchmarks is required for all salmon CUs in the Pacific Region of DFO. The Pacific Region identifies "Pacific Fisheries Reform' as a key priority in its '2006-2010 Pacific Region Implementation Plan' and lists as the first action, implementation of the WSP. Fraser Sockeye have been identified as one of the priorities for WSP CU benchmark development by the WSP Strategy 1 Steering Committee. Fraser Sockeye are a high profile species among British Columbia salmon stocks and, as such, have greater pressure to comply with the WSP to evaluate stock status. In addition, formal WSP stock status evaluations are conditions of certification for the Marine Stewardship

Council (MSC) for Fraser Sockeye Salmon identified in their 'Action Plan to Address Conditions for MSC Certification for British Columbia Sockeye Fisheries'. The deadline for lower benchmark development outlined in the MSC Action Plan is 'through December 2011'. Finally, WSP lower benchmarks for Fraser Sockeye will be used in the Fraser River Sockeye Spawning Initiative (FRSSI) to be used in simulation modelling to evaluate the performance of different management actions (escapement strategies) in relation to stock status prescribed by WSP benchmarks.

A WSP lower benchmark methodology paper has been recently approved through PSARC and published by the Canadian Science Advisory Secretariat (CSAS) (Holt et al. 2009). This paper evaluates four broad criteria for assessing stock status that includes recent abundances, recent temporal trends in abundance, distribution of spawners, and fishing mortality relative to stock productivity. Using multiple criteria to assess stock status is required, particularly in light of declining productivity observed for Fraser Sockeye stocks in recent years.

Subsequent to the development of these benchmarks, this request also includes the completion of a preliminary review of the stock status for each Fraser Sockeye CU. As described in the previous section, this will be an iterative process given all methods have not been fully assessed including evaluating stock status when multiple benchmarks are available.

Possibility of integrating this request with other requests in your sector or other sector's needs?

WSP lower benchmark priorities also include Barkley Sound Sockeye and Fraser River Chinook CUs. This request will be linked with work conducted by Science teams working on these other CUs. All three groups will provide leadership and guidance to the development of WSP lower benchmarks for the remaining CUs in the Region through the WSP Strategy 1 Steering Committee and Working Group. This work is being conducted by Regional and Area Science.

Intended Uses of the Advice, Potential Impacts of Advice within DFO, and on the Public:

Who will be the end user of the advice (e.g. DFO, another government agency or Industry?). What impact could the advice have on other sectors? Who from the Public will be impacted by the advice and to what extent?

Required directly by Stock Assessment and DFO Science to identify stock status for Fraser Sockeye stocks for provision of advice to internal and external groups.

Fraser Sockeye are a high profile species among British Columbia salmon stocks and, as such, have greater pressure to comply with the relatively new WSP to evaluate stock status. Formal WSP stock status evaluations are conditions of certification (for marketing Fraser Sockeye internationally) by the Marine Stewardship Council (MSC) for Fraser Sockeye Salmon; lower benchmark deadline as a condition of MSC certification is 'through December 2011.'

Information completed on Fraser Sockeye conservation unit stock status is also required to feed into the multi-stakeholder FRSSI process to evaluate performance of different management actions in relation to stock status prescribed by WSP lower benchmarks.

Date Advice Required:

Latest possible date to receive Science advice (dd/mm/yyyy): 05/01/2010

Rationale justifying this date: to have benchmarks in place to input into the FRISSI process and fishing season for 2011.

Funding: Specific funds may already have been identified to cover a given issue (e.g. SARCEP, Ocean Action Plan, etc.)
Source of funding: Expected amount:

Initiating Branch's Approval:

Approved by Initiating Director: ☐ Date (dd/mm/yyyy):

Name of initiating Director:

Send form via email attachment following instructions below:

Regional request: *Depending on the region, the coordinator of the Regional Centre for Science Advice or the Regional Director of Science will be the first contact person. Please contact the coordinator in your region to confirm the approach.*

National request: *At HQ, the Director of the Canadian Science Advisory Secretariat (Denis.Rivard@dfo-mpo.gc.ca) AND the Director General of the Ecosystem Science Directorate (Sylvain.Paradis@dfo-mpo.gc.ca) will be the first contact persons.*

PART 2: RESPONSE FROM SCIENCE

In the regions: to be filled by the Regional Centre for Science Advice.
At HQ: to be filled by the Canadian Science Advisory Secretariat in collaboration with the Directors of the Science program(s) of concern.

Criteria characterising the request:

- ☐ Science advice is requested (rather than just information)
- ☐ A sound basis of peer-reviewed information and advisory precedent already exists.
- ☐ Inclusiveness is an issue
- ☐ Advice on this specific issue has been provided in the past.
- ☐ Urgent request.
- ☐ DFO is not the final advisory body.
    - ☐ CEAA process
    - ☐ COSEWIC process
    - ☐ Other:

Constraints regarding the planning of a standard peer review/Workshop:

- ☐ External expertise required
- ☐ This is a scientifically controversial issue, i.e., consensus does *not* currently exist within DFO science.
- ☐ Extensive preparatory work is required.
- ☐ Determination of information availability is required (prior to provision of advice).
- ☐ Resources supporting this process are not available.
- ☐ Expected time needed for the preparatory work:
- ☐ Other (please specify):

Other criteria that could affect the choice of the process, the timelines, or the scale of the meeting:

- ☐ The response provided could be considered as a precedent that will affect other regions.
- ☐ The response corresponds to a new framework or will affect the framework currently in place.
- ☐ Expertise from other DFO regions is necessary.
- ☐ Other (please specify):

Recommendation regarding the advisory process and the timelines:

☐ Science Special Response Process (SSRP) ☐ Workshop ☐ Peer Review Meeting

Rationale justifying the choice of process:

Types of publications expected and if already known, number of report for each series:

☐ Science Advisory Report ( ) ☐ Research Document ( )
☐ Proceeding ( ) ☐ Science Response Report ( )
☐ Other:

Date Advice to be Provided:

☐ Date specified can be met.
☐ Date specified can NOT be met.

Alternate date, as agreed to by client Branch lead and Science lead (dd/mm/yyyy):

OR

☐ No Formal Response to be Provided by Science

Rationale:

- ☐ DFO Science Region does not have the expertise required.
- ☐ DFO Science Region does not have resources available at this time.
- ☐ The deadline can not be met.
- ☐ Not a natural science issue (e.g. socio-economic)
- ☐ Response to a similar question has been provided elsewhere:
  Reference:

Additional explanation:

Science Branch Lead:

Name: Telephone Number:
Email:

* Please contact Science Branch lead for additional details on this request.

Science Branch Approval:

Approved by Regional Director, Science (or their delegate authority):
☐ Date (dd/mm/yyyy):

Name of the person who approved the request:

Once part 2 completed, the form is sent via email attachment to the initiating Branch contact person.

PART 3: PLANNING OF THE ADVISORY PROCESS

Science Branch Approval:

Coordinator of the event:

Potential chair(s):

Suggested date (dd/mm/yyyy) / period for the meeting:

Need a preparatory meeting:

Leader of the Steering Committee: